# Aspects of the ecology of three mongoose species along a rural–urban landscape gradient of KwaZulu-Natal, South Africa

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## ABSTRACT

Many small carnivore species are of particular concern for conservation because of their elusive behaviour, diminutive size and crepuscular habits. Management and research tend to be challenging and thus is often limited. Several members of the Herpestidae family fall into this research dead zone, including the large grey mongoose (*Herpestes ichneumon*), slender mongoose (*Galerella sanguinea*), white-tailed mongoose (*Ichneumia albicauda*) and marsh or water mongoose (*Atilax paludinosus*). The distribution range and other aspects of the ecology of these and other such species, has remained vague for much of the sub-tropical regions of southern Africa because of the scarcity of comprehensive data. There is little understanding of the basic ecology of these mongoose species, and limited progress has been made in understanding the tolerance of these species within a changing environment. These species of mongoose appear to persist across KwaZulu-Natal's (KZN), South Africa, anthropogenically modified habitats (farmlands and urban). Small carnivore species, such as mongooses, can provide models of how medium-sized carnivores tolerate differing degrees of anthropogenic land-use change.

A global review of the current knowledge and research effort for urban mammalian mesocarnivores was conducted. In the last decade, there has been a dramatic increase in the number of studies that focus on medium-sized mammalian carnivores. Disproportionate levels of urban studies exist for mesocarnivores. Several species have been comprehensively studied in the United States of America and the United Kingdom. These two developed countries held more than half of all studies on urban mammalian mesocarnivores. Areas of the world that are undergoing rapid urbanisation have the most significant deficiency of research.

Across Africa, the spatial ecology of the Herpestidae family remains relatively poorly studied. We investigated how anthropogenic land-use change affects aspects of the ecology of three co-existing mongoose species (large grey, water and white-tailed) in the agricultural

setting of the KZN Midlands. Individuals of these three species of mongooses were trapped, collared and tracked using GPS-UHF (ultra-high frequency). The results presented show that the three focal species have different spatial requirements and movements from previous stated in literature. The natural habitat and farmland mosaic of the KZN Midlands are segregating the preferred habitats into small fragments. These niche fragments are intensely used by each species and allow them to co-exist in these anthropogenically modified environments. Besides these species' generalist nature, their behavioural plasticity may assist them in persisting in anthropogenically modified landscapes. In the absence of apex predatory control, an expansion in the range and population number of Herpestidae species is expected. This study provides crucial information on the spatio-temporal ecology of large grey, water and white-tailed mongoose in the KZN Midlands. The spatial movements of the three co-existing species in this study highlighted the variability that is present at a species and individual level. Further research is required to address the human–wildlife conflict that takes place locally. The urban aspect of the large grey, water and white-tailed mongoose ecology is also understudied, and we recommend further research be targeted.

The spatial ecology of water mongoose in the urban green space matrix of the Upper Highway Area of eThekwini, KZN, was subsequently investigated. This was to compare aspects of their ecology between fragmented natural farmland and urban mosaic using similar methods. Water mongooses occurred at a high density in the fragmented green Durban Metropolitan Open Space System (D'MOSS). Insight into the spatial ecology of urban water mongoose (home range, core area utilisation, overlap and habitat use) determined using GPS telemetry data from 14 water mongooses from July 2018 - October 2019. Water mongooses had constricted spatial movements that were highly dependent on natural refugia, and individuals intentionally avoided built-up areas of the urban matrix. The confined nature of these green spaces consequently resulted in home range sizes that were significantly smaller

than their farmland conspecifics. However, the species behavioural plasticity and generalist nature has allowed the species to occur at higher population densities in the green spaces of the urban matrix of the Upper Highway Area.

Furthermore, the effects of urban sprawl on the dietary ecology of water mongooses using faecal analyses was conducted. Water mongooses scat samples (n = 104) were opportunistically collected and with the aid of members of the Kloof Conservancy during the study. Urban water mongooses consume a diverse array of dietary items (9 categories), which is dominated by three categories (relative frequency of occurrence: crustaceans 35.7%, invertebrates 19.9%, small mammals 19.1% and other 25.2%). Seasonal variation in the diet only occurred for crustaceans and invertebrates with the other dietary categories consumed equally throughout the seasons. Additionally, it was demonstrated that urban water mongooses are supplementing their diets with anthropogenic waste (chicken bones, plastic particulates and cigarette butts). The broad diversity in dietary categories and supplementation of anthropogenic waste demonstrates the generalist opportunistic feeding behaviour and adaptability of the species in an urban matrix.

An online questionnaire survey was conducted to investigate socio-ecological attitudes and general perspectives towards mammalian mesocarnivores across a land-use gradient (rural–urban) from the uMgungundlovu to eThekwini Municipalities of KZN. Significant trends were assessed using the frequency of responses. The public held a range of different perspectives. Overall, respondents viewed mesocarnivores as non-threatening and vital for the environment. However, black-backed jackals (*Canis mesomelas*) were commonly identified as a problematic pest species, that subsequently are targets of predator control pressures by rural participants. The survey also identified that smaller, behaviourally flexible species (mongoose species and cape genet *Genetta tigrina*) were commonly sighted and are potentially increasing in both land-use types in the absence of competition and predatory control. Vehicle collisions

are the primary cause of mesocarnivore fatality reported by participants of the survey. The impact of mesocarnivores was perceived differently along a land–use gradient which links to levels of interaction. The study emphasises the importance of citizen science and community engagement when attempting to understand the drivers of human–wildlife interactions and potential mitigation strategies.

The present multifaceted study has improved our understanding of solitary members of the Herpestidae family and how anthropogenic changes affect them across a land-use gradient. Furthermore, the behavioural flexibility and adaptability of mongooses in enabling them to persist at differing degrees of anthropogenic pressure were evident. However, the size class of mammalian mesocarnivores remains understudied in Africa, and this is of concern in a rapidly developing region.

# PREFACE

The data described in this thesis were collected in KwaZulu-Natal, Republic of South Africa, from August 2016 to November 2019. Experimental work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, under the supervision of Prof Colleen T. Downs and co-supervision of Dr Tharmalingam Ramesh.

This thesis, submitted for the degree of Doctorate of Philosophy in Science in the College of Agriculture, Engineering and Science, University of KwaZulu-Natal, School of Life Sciences, Pietermaritzburg campus, represents original work by the author and has not otherwise been submitted in any form for any degree or diploma to any University. Where use has been made of the work of others, it is duly acknowledged in the text.

…….

Jarryd Peter Streicher

December 2020

I certify that the above statement is correct, and as the candidate's supervisor I have approved this thesis for submission.

……..

Professor Colleen T. Downs

Supervisor

December 2020

**COLLEGE OF AGRICULTURE, ENGINEERING AND SCIENCE**

**DECLARATION 1 - PLAGIARISM**

I, Jarryd Peter Streicher, declare that

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2. This thesis has not been submitted for any degree or examination at any other university.

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**DECLARATION 2 - PUBLICATIONS**

DETAILS OF CONTRIBUTION TO PUBLICATIONS that form part and/or include research presented in this thesis.

Publication 1

**JP Streicher, T Ramesh and CT Downs**

**Not all urban mesocarnivores are equal: A review of the global effort on urban mammalian research**

*Author contributions:*

JPS conceived paper with CTD and TR. CTD sourced funding. JPS collected and analysed data, and wrote the draft paper. CTD and TR contributed valuable comments to the manuscript.

Publication 2

**JP Streicher, T Ramesh and CT Downs**

**Home range and core area utilisation of three co-existing mongoose species: large grey, water and white-tailed in the fragmented landscape of the KwaZulu-Natal Midlands, South Africa**

Published

*Author contributions:*

JPS conceived paper with CTD and TR. CTD sourced funding. JPS collected and analysed data, and wrote the draft paper. CTD and TR contributed valuable comments to the manuscript.

Publication 3

**JP Streicher, T Ramesh and CT Downs**

**An African urban mesocarnivore: Navigating the urban matrix of Durban, South Africa**

Published

*Author contributions:*

JPS conceived paper with CTD and TR. CTD sourced funding. JPS collected and analysed data, and wrote the draft paper. CTD and TR contributed valuable comments to the manuscript.

Publication 4

**JP Streicher, T Ramesh and CT Downs**

**Community perceptions of mammalian mesocarnivores across a land-use gradient in KwaZulu-Natal, South Africa**

In review

*Author contributions:*

JPS conceived paper with CTD and TR. CTD sourced funding. JPS collected and analysed data, and wrote the draft paper. CTD and TR contributed valuable comments to the manuscript.

Publication 5

**JP Streicher, MB Streicher, T Ramesh and CT Downs**

**Diet of a generalist mammalian mesocarnivore species in an urban matrix**

In review

*Author contributions:*

JPS conceived paper with CTD and TR. CTD sourced funding. JPS collected and analysed data, and wrote the draft paper. CTD and TR contributed valuable comments to the manuscript.

Signed:  ……

Jarryd Peter Streicher

December 2020

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*Psalm 121*

Collared white-tailed (*Ichneumia albicauda*), water (*Atilax paludinosus*) and large grey (*Herpestes ichneumon*) mongoose in KwaZulu-Natal Midlands South Africa (Photo credit JP Streicher)

## TABLE OF CONTENTS

## FIGURES

## TABLES

# CHAPTER 1

# INTRODUCTION

## 1.1 Background and problem statement

Intensive anthropogenic land–use transformation (agriculture and urbanisation) is a significant threat to biodiversity globally (McKee et al. 2004; Ripple et al. 2014; Tilman et al. 2017). The anthropogenic process results in extensive fragmentation of natural habitats, which disrupts biological systems (Vitousek et al. 1997; Estes et al. 2011; Di Minin et al. 2016). Literature suggests a general inverse relationship between anthropogenic land–use change and overall loss of biodiversity as seen by a decrease in both fauna and fauna (Bennett and Saunders 2010; Pekin and Pijanowski 2012). However, highly adaptive fauna generalist species have been shown to persist and, in some circumstances, thrive under varying levels of anthropogenic pressure (Sih et al. 2011; Lowry et al. 2013). Certain species illustrate a resilience by exploiting human-modified landscapes by adjusting their natural behaviour (dietary, spatial and breeding ecology) (Lowry et al. 2013; Widdows and Downs 2015; Wong and Candolin 2015; Ngcobo et al. 2019a; Ngcobo et al. 2019b). It is of immense importance that we continue to generate knowledge on how animals responded to these anthropogenic pressures such that we can conserve wildlife biodiversity effectively.

Mammal species are often used as indicators of ecosystem health (Estes et al. 2011; Ripple et al. 2017). The range of size classes' and diversity of species provides a spectrum of the impacts of habitat fragmentation and human modification on mammals. Many small to medium-sized mammalian carnivore species are of particular concern for conservation and management (Prugh et al. 2009; Ripple et al. 2017). This size class of mammalian carnivores (mesocarnivores) tends to be challenging to study. Mesocarnivores are often characterised by

an elusive behaviour, diminutive size and crepuscular habits. Members of the Herpestidae family are suitable examples of mesocarnivores that display these behavioural traits.

*Herpestidae*

Mongooses belong to the family of Herpestidae, which was previously included in the family Viverridae (Skinner and Chimimba 2005). The Herpestidae now forms its own family which comprises 20 genera and 34 species that are distributed throughout Africa and Asia. Species of Herpestidae have been anthropogenically introduce into parts of Europe and isolated islands (Jennings and Veron 2020). Of this family, ~ 78% and ~ 40% of the genera and species respectively are found in the sub-region of southern Africa (Skinner and Chimimba 2005; Jennings and Veron 2020). The highly adaptive nature and plasticity in their behaviour of these mesocarnivores have enabled them to become abundant, and expand into various niches across the globe (Rasa 1987; Skinner and Chimimba 2005; Graw and Manser 2007; Kern and Radford 2014; Ramesh and Downs 2014). The members of the family are characterised by a cohort of morphometrics traits. These include long tails, short legs, and sharp muzzles (Skinner and Chimimba 2005). The Herpestidae exhibit a range of behavioural and morphological diversity, which has allowed the species to diversify in Africa.

The Herpestidae family is relatively diverse in southern Africa; however, there is limited research and literature on them (Skinner and Chimimba 2005). Their elusive behaviour and diminutive size have proven a challenging aspect to the study of the family. The above-mentioned factors have restricted the number of studies on the family in southern Africa, therefore Herpestidae have not been studied at the same detail as other southern Africa carnivore species. Only the disease, community, spatial and behaviour ecology of meerkat (*Suricata suricata*) have been extensively studied because of the species dynamic social structure (Bateman et al. 2013). Their social dynamic group behaviour has resulted in extensive

research being invested by institutes from developed countries. There is an evident gap in the literature concerning other species belonging to Herpestidae, partially their spatial behaviour (home range and habitat use). In the late 1980s, Maddock and Perrin conducted a baseline study on the spatial ecology of this family in southern Africa (Maddock 1988; Maddock and Perrin 1993). There has been a limited number of studies on the spatial ecology of Herpestidae in Africa (Table 1.1). Small carnivores such as mongoose could provide excellent models on how mesocarnivores cope with anthropogenically modified landscapes (Ramesh and Downs 2014).

Three mongoose species were the focus of this study: large grey, water and white-tailed mongoose species (*Herpestes ichneumon, Atilax paludinosus* and *Ichneumia albicauda*). These members of the family Herpestidae are characterised by distinctive behaviour and morphology (Table 1.2). These adaptations to their environment have allowed each species to occupy a distinct niche and to co-exist with limited niche overlap (Maddock, 1988).

*Conservation status*

Recently, the International Union for the Conservation of Nature (IUCN) classified the large grey, water and white-tailed mongoose species as species of least concern on the Red List of Threatened Species (Do Linh San 2015; Do Linh San et al. 2015; Do Linh San et al. 2016). These species have an extensive range of distribution throughout most of Africa (Figure 1.1). Additionally, the large grey mongoose non-native range includes parts of the southern peninsular of Turkey, Portugal and Spain after their introduction to these countries by humans (Do Linh San 2015; Do Linh San et al. 2015; Do Linh San et al. 2016). If suitable habitat is available, these three species of mongoose are generally considered to be present and common.

**Table 1. 1** A summary of studies focusing on the spatial ecology of species in the Herpestidae family in Africa. (Note: Studies from this thesis are omitted)

| Study Species | Scientific name | Reference |
|---|---|---|
| Yellow mongoose | *Cynictis penicillata* | Cavallini 1993; Cavallini and Nel 1995; Waterman and Roth 2007; Blaum et al. 2008; Le Roux et al. 2008; Cronk and Pillay 2019; Cronk and Pillay 2020, Cronk and Pillay 2021 |
| Water mongoose | *Atilax paludinosus* | Maddock 1988; Maddock and Perrin 1993; Ray 1997; Ziegler et al. 2002; Burton et al. 2012; Ramesh and Downs 2015 |
| Slender mongoose | *Galerella sanguinea* | Maddock 1988; Maddock and Perrin 1993; Waser et al. 1995; Ziegler et al. 2002; Martinoli et al. 2006; Blaum et al. 2008; Durant et al. 2010; Pettorelli et al. 2010; Ramesh and Downs 2014; Graw et al. 2016; Cronk and Pillay 2020 |
| Large grey / Egyptian mongoose | *Herpestes ichneumon* | Maddock 1988; Maddock and Perrin 1993; Ziegler et al. 2002; Ramesh and Downs 2015 |
| White-tailed mongoose | *Ichneumia albicauda* | Ikeda et al. 1982; Ikeda et al. 1983; Waser et al. 1995; Ziegler et al. 2002; Admasu et al. 2004; Martinoli et al. 2006; Durant et al. 2010; Pettorelli et al. 2010; Ramesh and Downs 2015 |
| Gambian mongoose | *Mungos gambianus* | Ziegler et al. 2002; Burton et al. 2012) |
| Jackson's mongoose | *Bdeogale jacksoni* | De Luca and Rovero 2006 |
| Dwarf mongoose | *Helogale parvula* | Creel and Rabenold 1994; Waser et al. 1995; Durant et al. 2010 |
| Black mongoose | *Galerella nigrata* | Rathbun et al. 2005; Rathbun and Cowley 2008) |
| Banded mongoose | *Mungos mungo* | (Maddock 1988; Maddock and Perrin 1993; Waser et al. 1995; Gilchrist and Otali 2002; Martinoli et al. 2006; Durant et al. 2010 |
| Cape grey mongoose | *Galerella pulverulenta* | Cavallini and Nel 1990; Cavallini and Nel 1995; Waser et al. 1995 |
| Bushy-tailed mongoose | *Bdeogale crassicauda* | Martinoli et al. 2006; Pettorelli et al. 2010 |
| Long-nosed mongoose | *Herpestes naso* | Ray 1997 |
| Meerkat | *Suricata suricata* | Manser and Bell 2004; Bateman et al. 2015 |

**Table 1. 2** Summary of the three mongoose species biology in the present study.

| | Large grey | | Water | | White-tailed | |
|---|---|---|---|---|---|---|
| | Male | Female | Male | Female | Male | Female |
| Body mass (kg) Mean and range | 3.3 (2.6–4.1) | 3 (2.2–4.1) | 3.2 (2.0–4.0) | 3 (2.0–4.1) | 4.5 (3.6–5.2) | 4.1 (3.6–4.9) |
| Social organisation | Solitary/breeding pairs | | Solitary/breeding pairs | | Solitary/family groups | |
| Home range (ha) | 300–450 | | 150–230 | | 110–541 | |
| Diel activity pattern | Diurnal | | Nocturnal | | Nocturnal | |
| Living requirements | Moist grasslands, Riparian conditions | | Water dominated systems (rivers, streams, waters, etc.) | | Well-watered Woodland and Bushlands | |
| Diet | Generalist carnivore. Occasionally feeds on vegetation | | Opportunistic omnivore | | Insectivore, Opportunistic omnivore | |
| Pelage colour | Ranges from grey to reddish-brown, ticked with brown or yellow flecks | | Black or reddish-brown or rusty | | Jet black limbs, grey pelt and bushy whitetail | |
| Mean litter size | 3.3 | | 2.5 | | 1.4 | |
| Predators | Raptors and larger carnivores | | Jackal, snakes and raptors | | Jackal, snakes and raptors | |
| Threats | Habitat fragmentation, land-use change, poisoning | | Habitat transformation, water pollution and slitting | | Habitat transformation, Habitat fragmentation, none target killing | |

(Rowe-Rowe 1992; Admasu et al. 2004; Skinner and Chimimba 2005)

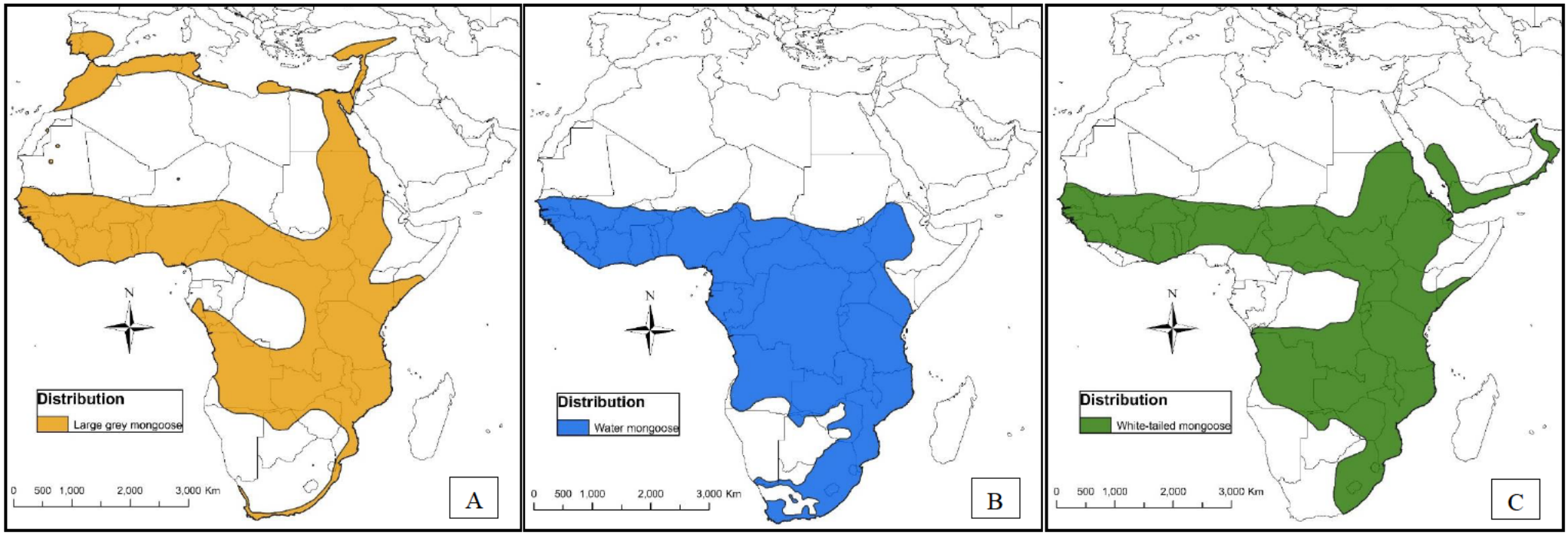


**Figure 1. 1.** The global extant distribution for three species of solitary mongooses: A- large grey mongoose (*Herpestes ichneumon*) (Do Linh San et al. 2016), B- water mongoose (*Atilax paludinosus*) (Do Linh San et al. 2015) and C- white-tailed mongoose (*Ichneumia albicauda*) (Do Linh San 2015).

Human threats to these mongoose species include indirect poisoning, use in the bushmeat trade, and targeted persecution for use in traditional medicines (Skinner and Chimimba 2005; Do Linh San 2015; Do Linh San et al. 2015; Do Linh San et al. 2016). Although there are no major threats imposed on the subject species, human encroachment and habitat fragmentation is invariably constraining, and thus a concern for conserving these species (Maddock 1988; Angelici et al. 1999). The water mongoose, in particular, is dependent on riverine and freshwater habitats for dietary resources and sheltering (Rowe-Rowe 1992; Skinner and Chimimba 2005). However, human encroachment and anthropogenic land-use change threaten their existence. The conversion of wetlands and swamplands in eastern African into arable farm areas has been indicated as a threat (Andama 2000).

Furthermore, the use of pesticides on agriculture crops, which ultimately runs off into water systems, endangers the species. Literature suggests that there are opposing views on the importance of Herpestidae members. Globally, mongoose species are viewed as vermin and pests because of their undesirable impact on farmlands and destructive consequences when introduced to islands (Maddock 1988; Hays and Conant 2007; Ramesh and Downs 2014). However, in regions of North Africa, the large grey mongoose is protected by the community because of its valued ability in being a natural predator to snakes (Maddock 1997).

Although species belonging to the Herpestidae family are widely distributed throughout Africa, their range distribution has remained vague for much of the sub-tropical regions of southern Africa. On the whole, there persists an evident lack of detail in which the family has been studied compared with other families of carnivores in Africa. Presently there is a basic understanding of the ecological tolerance of several mongoose species including the large grey mongoose, slender mongoose (*Galerella sanguinea*), white-tailed mongoose and water mongoose otherwise known as marsh mongoose. All species to have a widespread distribution range across a number of habitats (rural–urban) in KwaZulu-Natal Province, South Africa

(Skinner and Chimimba 2005; Ramesh and Downs 2015). Small carnivore species such as the mongooses can potentially provide excellent models of how carnivores persist with different levels human-modified land-use change. However, relatively little ecological work has been conducted since the previous research of Rowe-Rowe (1992), Maddock (1988) and Maddock and Perrin (1993) in KwaZulu-Natal. Mongooses are often regarded as vermin by local farm owners, because of the negative impacts on farming stock (especially domestic chickens *Gallus* spp. and egg supply), leading to issues of human–wildlife interactions.

*GPS telemetry*

Advances in Global Positioning System (GPS) telemetry has allowed increased access to understanding the fine-scale spatial-temporal ecology of wildlife (Ropert-Coudert and Wilson 2005; Cagnacci et al. 2010; Kays et al. 2015). Developments in tracking technologies have created expanding opportunities for accurate and continual ecological monitoring of species (Cooke et al. 2004; Kays et al. 2015). GPS tracking provides notable advantages contrasted to ARGOS satellite telemetry, radio telemetry (UHF or VHF) and light-based geolocation (Cagnacci et al. 2010; Hebblewhite and Haydon 2010; Tomkiewicz et al. 2010). These benefits include active, continual tracking of individuals for longer durations regardless of environmental conditions (Arthur and Schwartz 1999), removal of bias and or human observational localities, increased spatial accuracy and increased storage memory to record large sets of observational recordings (Cagnacci et al. 2010; Kays et al. 2015). These advantages become critical when attempting to track small, crepuscular and elusive animals in relatively hostile environments (Hulbert and French 2001; Recio et al. 2011). However, present drawbacks of using GPS loggers are their expense, issues of autocorrelation or independence in location data and difficulties associated with trapping elusive wildlife species (Cooke et al. 2004; Walter et al. 2011). These factors heavily influence the practicality of a successful study,

requiring adequate monetary resources and novel trapping techniques. Despite this, the significance of tracking a single individual of unestablished spatial ecology for an extended period can provide important ecological data that was previously undocumented (Kays et al. 2015). These novel ecological data may have important conservation, biodiversity and behavioural implications, particular in a rapidly changing anthropogenically altered climatic and physical world (Cooke 2008; Kays et al. 2015).

## 1.2 Aims and objectives

*Aims*

This project aimed to fill evident knowledge gaps present in the ecology (habitat use, home ranges, movements) of three mongoose species: large grey, water mongoose and white-tailed mongoose found in anthropogenically modified landscapes in KwaZulu-Natal. The research aims to establish conservation requirements of these three species of mongooses through an improved ecological understanding across a land-use gradient to improve conservation measures for mongooses. The study took place in the fragmented natural habitat and farmland mosaic of the in KwaZulu-Natal Midlands to the urban Metropole of eThekwini, KwaZulu-Natal, South Africa. Additionally, we aimed to understand the socio-ecological factors and attitudes towards mammalian mesocarnivores across this land-use gradient. Finally, we aimed to determine the dietary ecology of water mongooses in the urban setting of residential Kloof, eThekwini, KwaZulu-Natal.

*Objectives*

The objectives of this project on three mongoose species across an anthropogenically modified landscape in KwaZulu-Natal were as follows:

- To highlight the present research gaps and bias in urban mesocarnivore studies globally with particular emphasis on Africa.
- To determine the home range and core area utilisation of three co-existing mongoose species: large grey, water and white-tailed in natural and farmland land-uses in the KwaZulu-Natal Midlands.
- To determine the spatial ecology of water mongoose, in a fragmented urban matrix of the Upper Highway Area, eThekwini, KwaZulu-Natal.
- To determine the socio-ecological factors and attitudes towards mammalian mesocarnivores, of the public across a land-use gradient in KwaZulu-Natal.
- To determine the diet of water mongooses in the urban setting of residential Kloof, eThekwini, KwaZulu-Natal.
- To make management recommendations for the conservation of three mongoose species: large grey, water and white-tailed mongoose across an anthropogenic land-use gradient from natural and farmlands of the KwaZulu-Natal Midlands to the urban matrix of eThekwini.

### 1.3 Structure of the thesis

This thesis is structured with a brief introduction followed by a stand-alone review, and data chapters that are prepared and formatted for submission (some are already published or in review) to international peer-reviewed journals. Some repetition was, therefore, unavoidable. A final chapter concludes these. The chapters are:

• Chapter 1: Introduction

• Chapter 2: Not all urban mesocarnivores are equal: A review of the global effort on urban mammalian research

• Chapter 3: Home range and core area utilisation of three co-existing mongoose species: large grey, water and white-tailed in the fragmented landscape of the KwaZulu-Natal Midlands, South Africa

• Chapter 4: An African urban mesocarnivore: Navigating the urban matrix of Durban, South Africa

• Chapter 5: Community perceptions of mammalian mesocarnivores across a land-use gradient in KwaZulu-Natal, South Africa

• Chapter 6: Diet of a generalist mammalian mesocarnivore species in an urban matrix

• Chapter 7: Conclusions.

### 1.4 References


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**CHAPTER 2**

# Not all urban mesocarnivores are equal: A review of the global effort on urban mammalian research

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**Running header:** African urban mesocarnivore review

## 2.1 Abstract

Human population increase and anthropogenic land-use changes are impacting biodiversity globally. Information on the impacts on urban wildlife is becoming increasingly apparent. Despite an increasing trend in urban wildlife studies, a systematic assessment of mammalian mesocarnivores in urban areas is lacking. Using a systematic review process, we evaluated all urban studies conducted on mammalian carnivores that are published in internationally peer-reviewed journals. We assessed urban wildlife publications from 1970–2020 to quantify trends over time regarding family and geographical focus of publications across the globe. Urban mammalian studies have increased progressively through the decades, with 85% of all studies conducted in the last two decades (2001–2010 = 27%, 2011–2020 = 58%). Geographically we found that mesocarnivore studies were disproportionately conducted with 77.3 % of all studies in North America (44.7%) and Europe (32.6%) with Australia, Africa, Asia, South America and global reviews accounting for the remaining studies. Furthermore, the United States of America (47.9%) and the United Kingdom (13.6%) contributed to 61.5% of all studies focused on a single mesocarnivore species or multiple species. We found eight carnivores family represented in urban studies consisting of 61 species. Two canid species *Vulpes vulpes* (23.2%) and *Canis latrans* (17.5%) accounted for 40.7% of all studies conducted on urban mesocarnivores. Critical gaps in research for urban mammalian carnivores in rapidly urbanising areas in Asia, Africa, and South America were evident with species in these areas understudied.

***Keywords***: Literature review, Wildlife, Urban ecology, Mammal, Mesocarnivore

## 2.2 Introduction

The conversion of natural/semi-natural and agricultural landscapes into urbanised areas is the foremost process of land-use change globally (Pickett et al. 2001; Šálek et al. 2015). Urbanisation is projected to expand at a disproportionate rate worldwide during the 21st century to accommodate the rapid increase in urban population (Montgomery 2008; Seto et al. 2011). The continual development of urban areas has critical implications for biodiversity and its conservation as a direct result of disturbance, fragmentation, and modification of natural habitats (Haddad et al. 2015; Lepczyk et al. 2017; Marzluff 2001). The accelerated rate of urbanisation globally has resulted in a paralleled, and much needed, increase in wildlife research in urban areas (McKinney 2008; Niemelä 2014; Pickett et al. 2008; Ramalho & Hobbs 2012). Historically, there have been isolated attempts to investigate urban wildlife before the 1970s (Gehrt et al. 2010). Modern-day urban wildlife ecology has subsequently expanded its footprint as a multi-disciplinary and internationally recognised field (Magle et al. 2012). Urban ecosystems are increasingly recognised as essential drivers in the protection of local biodiversity either at a species-specific level or through the preservation of natural and human-managed urban green spaces (corridors or steppingstone patches) within the urban mosaic landscape (Alexander et al. 2019a, 2019b, 2019c; Beninde et al. 2015; Lepczyk et al. 2017; Maseko et al. 2020; Zungu et al. 2020a; Zungu et al. 2020b). Besides the conservation of local biodiversity and ecosystem services, "urban greening" results in numerous human benefits (air quality, hydrology, carbon sequestration, pollination and human quality of life) (Kabisch et al. 2017; Lepczyk et al. 2017; Magle et al. 2012). Natural and human-managed green spaces in urban areas allow residents to interact with the natural environment and assists in developing appreciation for nature in an otherwise concrete world (Lafortezza et al. 2018). Access to green spaces in cities provides biodiversity conservationists with the ability to interact with the city denizens through strategies, awareness programmes and other recreation-based activities

(Aronson et al. 2017). The inescapable decline of natural habitats left available for conserving wildlife necessitates the preservation of urban conservation areas, even pocket communities (Dearborn & Kark 2010). Preserving these remaining communities is becoming increasingly urgent.

Studies illustrate a generally inverse relationship between land-use change and species diversity, in both spatial distribution and overall abundance (Di Marco et al. 2014; Sala et al. 2000). Many vertebrates avoid human-dominated areas (urban, residential and industrial areas) because of the disruption caused in ecosystem functions and availability of resources (dietary, habitat, shelter prey and breeding sites) (Kark et al. 2007; Lowry et al. 2013). There are, however, exceptions where numerous mammalian carnivore species have thrived in heavily human-modified landscapes. These species have been documented to show behavioural plasticity (especially in spatial, activity, dietary and interaction behaviours) to exploit anthropogenic resources maximally, and thereby increase their population size and densities (Murray & Clair 2015; Newsome et al. 2015; Oriol-Cotterill et al. 2015). Most of these studies are confined to Europe and North America (Magle et al. 2012). This literature shows how small to large mammalian carnivores such as the coyote (*Canis latrans*), red fox (*Vulpes vulpes*), dingo (*Canis lupus dingo*), black bear (*Ursus americanus*), cape genet (*Genetta tigrina*) often become accustomed and dependent on anthropogenic food resources associated with human-dominated landscapes (Bateman & Fleming 2012; Glen & Dickman 2005; Lewis et al. 2015; Murray & Clair 2015; Prugh et al. 2009; Ritchie & Johnson 2009; Widdows & Downs 2015, 2016). In some cases, species have altered their activity pattern in switching from previously diurnal behaviour to crepuscular nocturnal activity or vice versa (Murray & Clair 2015; Widdows & Downs 2018). This modification has allowed numerous mammalian carnivores to become the uncontended exploiters (synoptic species) of an open and resource-abundant niche which facilitates their expansion (Brashares et al. 2010; Prugh et al. 2009; Roemer et al. 2009).

Notably, these mammalian exploiters of the urban ecosystems are characterised by a suite of shared traits. Urban visitors and dwellers tend to be small to medium in size and exhibit high reproductive potential (r-select species) (Santini et al. 2019). Francis and Chadwick (2012) defined synurbic as a species which colonises or is found within urban ecosystems with higher population density than rural areas. Synurbic carnivore species exhibit diverse dietary breadth and exhibit behaviourally flexibility (Gehrt et al. 2010; Sol et al. 2008). Lastly synurbic carnivore species need to be tolerant of humans and in turn require human tolerances (Santini et al. 2019). Although there are exceptions (e.g. black bears: Lewis et al. 2015) in terms of size and reproductive potential, successful mammalian carnivores are suggested to possess these common traits.

Research focused on urban exploiters has been fundamental to establishing conservation strategies and collaboration initiatives, and in broadening the scope of understanding of the differential relationship present between biodiversity and land-use intensity. However, this focus of research is sparse in developing countries and has mostly been side-lined. There is, therefore, a significant gap in our understanding of the effects that rapid human population expansion is having on wildlife species in developing regions. Despite the dramatic global increase in studies that have focused on urban carnivore ecology, there has not been a comprehensive review of mammalian mesocarnivore species in urban areas that also includes developing countries, especially those from the global south. We need a "stocktake" of current knowledge to assess the research effort and subsequent gaps in knowledge in urban mesocarnivores studies to understand trends in mesocarnivore responses to urbanisation. To assess this research effort, we reviewed all mammalian mesocarnivore literature across a broad range of fields which had been conducted in urban areas. We defined urban mammalian mesocarnivore in line with the Prugh et al. (2009) definition of a mesopredator as of any naturally occurring midranking mammalian carnivores in the food web, regardless of its size

that inhabits an urban landscape. We focused our study on the major regions of the world to ascertain the interest in urban mesocarnivore research. We show that research effort is not uniform, neither globally nor in target species. The apparent bias in research may have crucial implications on how to progress in furthering our understanding of mesocarnivore conservation globally.

**2.3 Methods**

We carried out a comprehensive literature search to quantify the available research on urban mammalian mesocarnivores. We compiled data from only internationally peer-reviewed published journals. Article titles were found using the Web of Science and Google Scholar online search engine databases using the search phrase "carnivore" with the following keyword "*" notation: urban, suburban, city, periurban, exurban, residential. Publications were incorporated in the review if they represented wildlife-based research (non-domestic animals) located in areas of human-dominated landscapes (non-agricultural). Exurban and the rural regions represent areas outside of the immediate urban landscape (Theobald 2001); however, they were included in the analyses because of their association with the urban habitat (Magle et al. 2012). Review papers were included in the analyses, while editorials, letters, comments, and book reviews were excluded (Magle et al. 2012)

For the Web of Science and Google Scholar outputs, the results were refined by countries belonging to major global regions (North America, Europe, Asia, Australia, South America and Africa). The search was further refined by using the Web of Science Core Collection with the combination of specific terrestrial families of carnivorous terrestrial mammals Canidae, Felidae, Procyonidae, Mustelidae, Mephitidae, Herpestidae and Viverridae with the "*" notation. Although Didelphidae, belong to marsupials we considered that family in addition to the Carnivora. Journal articles were subsequently separated by species of interest

and tabulated by global region and country in which the study was performed (Supplementary Information Table S2.1). Additional studies were incorporated into the final summary table by referral.

Furthermore, publications were evaluated for their primary research topics. The classification process was not limited to a single research topic and could comprise multiple topics. The ten primary research topics were: 1. Animal behaviour: spatial movement (dispersal, home range, habitat) intraspecific interactions (mating and conflicts), alterations in activities and diet, 2. Community ecology: interspecific interaction between two or more species, 3. Conservation: studies focused on endangered or threatened species, 4. Genetics: population genetics, 5. Human dimension: citizen science and survey questionnaires, 6. Human–wildlife conflict: wildlife attacks, economic damage, domestic pet interactions, vehicle collisions, 7. Population ecology: demographic levels and population change, 8. Wildlife diseases: zoonotic and wildlife diseases, parasite and transmission, 9. Wildlife management: population control, baiting methods and sterilisation and 10. Wildlife toxicology: anthropogenically derived blood toxins, poisons and heavy metals. We collated all these data and used descriptive statistics to show trends.

### 2.4 Results

We reviewed 3,888 total publications generated by our keyword search within the Web of Science and Google Scholar from 1970–2020 for urban mesocarnivores. A total of 503 peer-reviewed journal publications met the requirements we set for this review. Before 1991 a total of 27 (5.4%) studies focused on urban mesocarnivores (1971–1980 = 5 and 1981–1990 = 22). In the proceeding decades, there was a dramatic increase in the number of publications focusing on urban mesocarnivores (1991–2000 = 50 and 2001–2010 = 137). Between 2011–2020, urban

mesocarnivore studies have more than doubled as compared with the previous decade (Fig. 2.1). The present decade contributed 289 internationally peer-reviewed journal articles (57.5%).

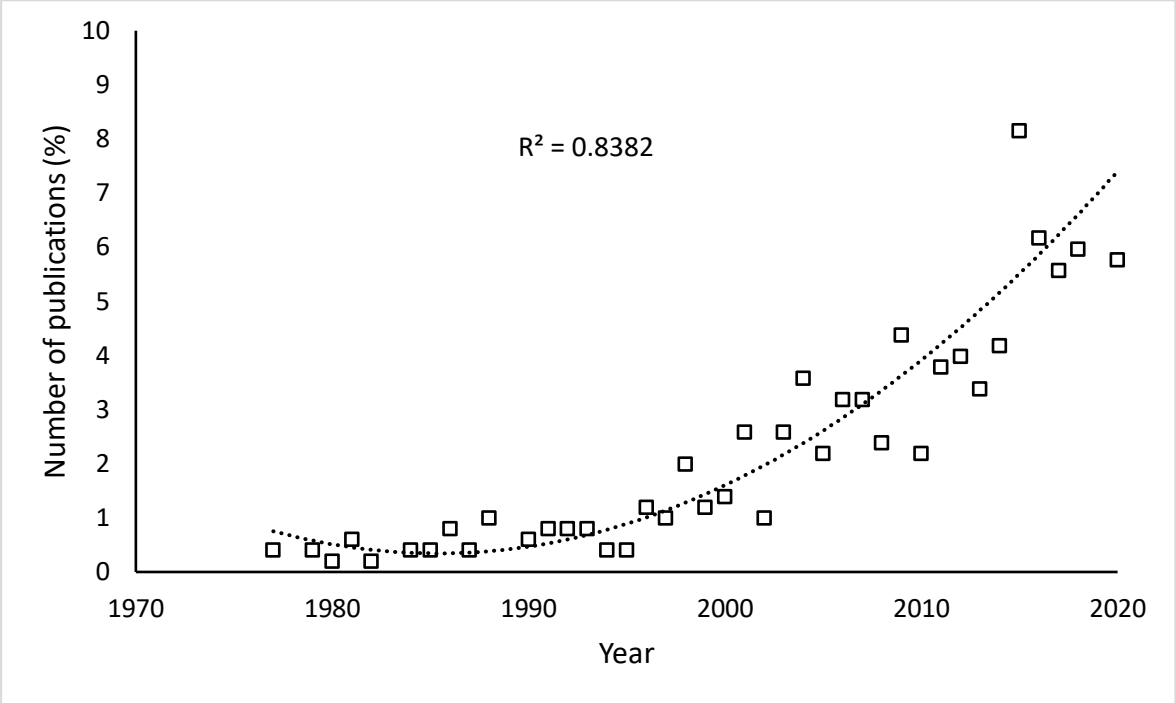


**Figure 2.1.** Percentage of urban wildlife publications focusing on mesocarnivores from 1977 to 2020. Data are presented as the percentage of the total number of publications. A second-order polynomial function trendline indicates increasing publication volume (y = 0.0059x2 - 23.371x + 23200).

*Geographic focus*

Urban mammalian mesocarnivore studies were disproportionately conducted on a global scale. The majority of wildlife studies were conducted in North America (225/503, 44.7%), and Europe (164/503, 32.6%). Fewer studies occurred in Asia (36/503, 7.2%), South America (24/503, 4.8%), Africa (21/503, 4.2%), and Australia (18/503, 3.6%). Global reviews, which were conducted on more than one continent, were rare (15/503, 3.0%) (Fig. 2.2; Appendix S2.1). The countries from each region which produced the greatest research output were: United States of America (North America) 203 studies (47.9%), United Kingdom (Europe) 66 studies

(13.6%), Japan (Asia) 29 studies (6.0%), South Africa (Africa) 19 studies (3.9%) and Brazil (South America) 13 studies (2.6%) (Fig. 2.3; Appendix S2.1).

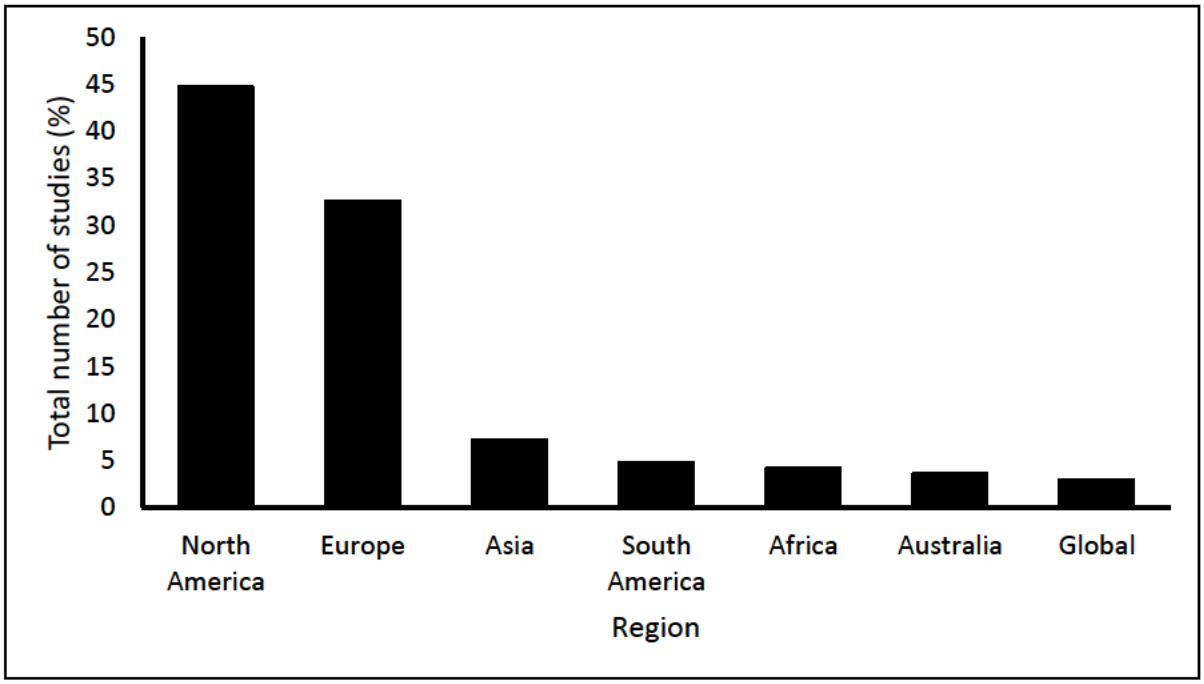


**Figure 2.2.** The percentage of urban mammalian mesocarnivore studies for each global region published in internationally peer-reviewed journal papers from 1970–2020.

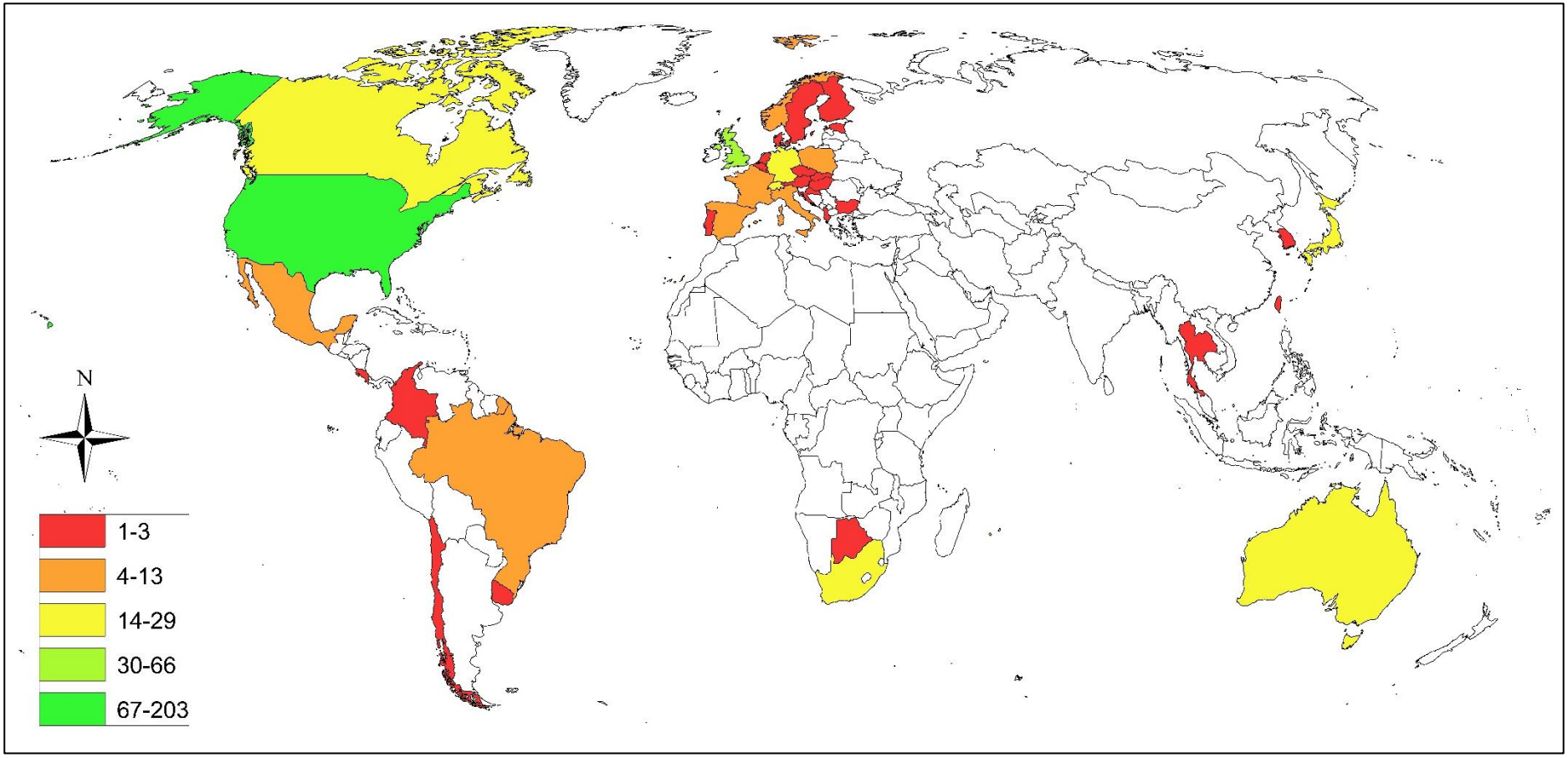


**Figure 2.3.** The number of urban mammalian mesocarnivore studies for each country which have been published in peer-reviewed journal papers from 1970 – 2020.

*Family and species focus*

A total of eight carnivore taxa consisting of 61 species were represented in this study. The family most frequently studied was Canidae (50.8%), followed by Mustelidae (17.6%), Procyonidae (9.8%), Felidae (8.7%), Mephitidae (4.7%), Didelphidae (3.4%), Viverridae (2.6%) and the Herpestidae (2.4%) was the least studied (Fig. 2.4). The Mustelidae was represented by the most diverse number of species (24), followed by Canidae (12), Herpestidae (7), Felidae (5), Procyonidae (4), Viverridae (4), Didelphidae (3) and Mephitidae (2). Within each family, a single species often dominated urban research output above others. For Canidae - *Vulpes vulpes*, Mustelidae - *Meles meles*, Procyonidae - *Procyon lotor*, Felidae - *Lynx rufus*, Mephitidae - *Mephitis mephitis*, Didelphidae - *Virginia opossum*, Viverridae - *Genetta genetta* and *Genetta tigrina* and Herpestidae - *Cynictis penicillata* (Table 2.1).

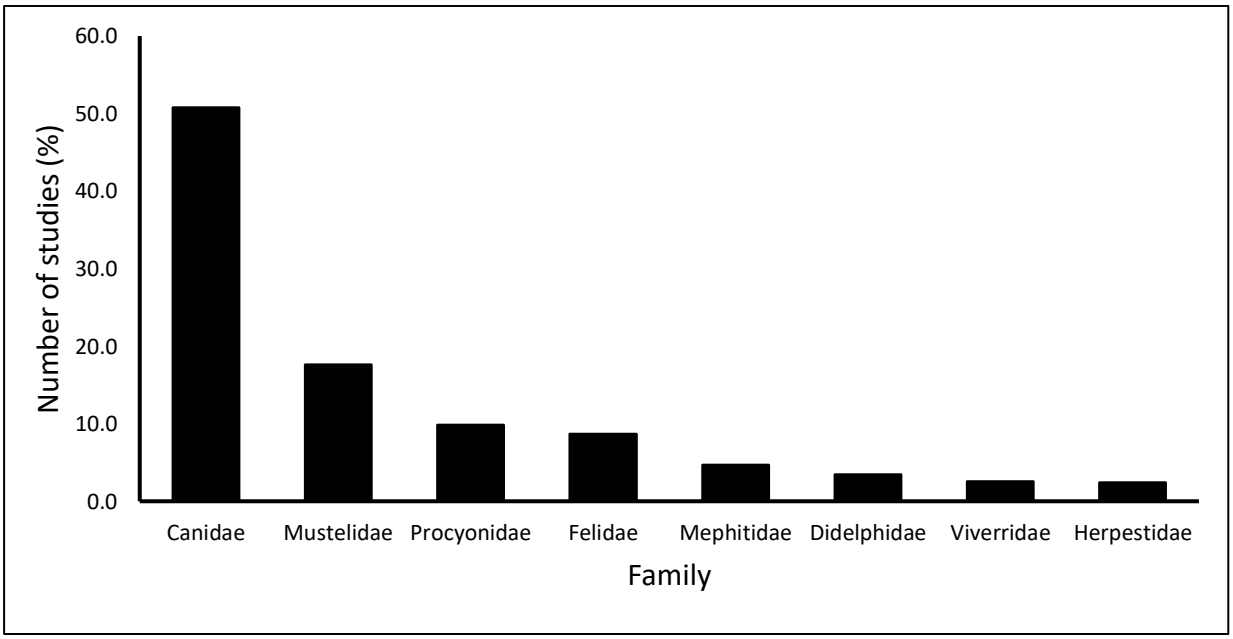


**Figure 2.4.** The percentage of studies on urban mammalian mesocarnivore families that have been published in peer-reviewed journal papers from 1970–2020.

**Table 2.1.** Total representation and percentage of urban mesocarnivore species that have been published in peer-reviewed journal papers.

| Family | Species | Number of studies | Total percentage of studies (%) |
|---|---|---|---|
| Canidae | *Canis latrans* | 123 | 17.5 |
| | *Canis lupus dingo* | 4 | 0.6 |
| | *Canis mesomelas* | 2 | 0.3 |
| | *Cerdocyon thous* | 4 | 0.6 |
| | *Chrysocyon brachyurus* | 2 | 0.3 |
| | *Nyctereutes procyonoides* | 19 | 2.7 |
| | *Urocyon cinereoargenteus* | 22 | 3.1 |
| | *Urocyon littoralis* | 2 | 0.3 |
| | *Vulpes chama* | 1 | 0.1 |
| | *Vulpes lagopus* | 2 | 0.3 |
| | *Vulpes macrotis* | 13 | 1.8 |
| | *Vulpes vulpes* | 163 | 23.2 |
| Mustelidae | *Aonyx capensis* | 5 | 0.7 |
| | *Gulo gulo* | 3 | 0.4 |
| | *Hydrictis maculicollis* | 1 | 0.1 |
| | *Ictonyx striatus* | 1 | 0.1 |
| | *Lontra canadensis* | 5 | 0.7 |
| | *Lontra provocax* | 1 | 0.1 |
| | *Lutra lutra* | 7 | 1.0 |
| | *Lutrogale perspicillata* | 1 | 0.1 |
| | *Martes flavigula* | 1 | 0.1 |
| | *Martes foina* | 23 | 3.3 |
| | *Martes martes* | 8 | 1.1 |
| | *Meles meles* | 29 | 4.1 |
| | *Meles meles anakuma* | 3 | 0.4 |
| | *Mellivora capensis* | 2 | 0.3 |
| | *Melogale moschata* | 2 | 0.3 |
| | *Mustela erminea* | 1 | 0.1 |
| | *Mustela frenata* | 6 | 0.9 |
| | *Mustela itatsi* | 3 | 0.4 |
| | *Mustela nivalis* | 4 | 0.6 |
| | *Mustela putorius* | 5 | 0.7 |
| | *Mustela sibirica* | 1 | 0.1 |
| | *Neovison vison* | 7 | 1.0 |
| | *Pekania pennanti* | 2 | 0.3 |
| | *Taxidea taxus* | 3 | 0.4 |

| | | | |
|---|---|---|---|
| Procyonidae | *Bassariscus astutus* | 4 | 0.6 |
| | *Nasua narica* | 2 | 0.3 |
| | *Nasua nasua* | 9 | 1.3 |
| | *Procyon lotor* | 54 | 7.7 |
| Felidae | *Caracal caracal* | 5 | 0.7 |
| | *Felis silvestris* | 2 | 0.3 |
| | *Lynx lynx* | 5 | 0.7 |
| | *Lynx rufus* | 48 | 6.8 |
| | *Neofelis nebulosa* | 1 | 0.1 |
| Mephitidae | *Mephitis mephitis* | 31 | 4.4 |
| | *Spilogale gracilis* | 2 | 0.3 |
| Didelphidae | *Didelphis marsupialis* | 2 | 0.3 |
| | *Philander opossum* | 2 | 0.3 |
| | *Virginia opossum* | 20 | 2.8 |
| Viverridae | *Genetta genetta* | 6 | 0.9 |
| | *Genetta tigrina* | 6 | 0.9 |
| | *Paguma larvata* | 5 | 0.7 |
| | *Viverricula indica* | 1 | 0.1 |
| Herpestidae | *Atilax paludinosus* | 2 | 0.3 |
| | *Cynictis penicillata* | 5 | 0.7 |
| | *Galerella pulverulenta* | 1 | 0.1 |
| | *Galerella sanguinea* | 4 | 0.6 |
| | *Herpestes ichneumon* | 2 | 0.3 |
| | *Herpestes urva* | 1 | 0.1 |
| | *Mungos mungo* | 2 | 0.3 |

*Scientific study focus*

The most frequently represented scientific topics for mammalian mesocarnivore research was animal behaviour (140/503, 27.9%) and wildlife diseases (70/503, 13.9%) (Fig. 2.5). Other single topic studies were less common: genetics (11/503, 2.2%), human dimensions (9/503, 1.8%), wildlife toxicology (7/503, 1.4%), wildlife management (7/503, 1.4%), community ecology (6/503, 1.2%), population ecology (5/503, 1.0%), human–wildlife conflict (4/503, 0.8%) and conservation (2/503, 0.4%). Common topic combinations were animal behaviour and population ecology (25/503, 5%) and animal behaviour and community ecology (25/503, 5%). The remaining topics were represented by less than five of the total publications on mammalian mesocarnivores.

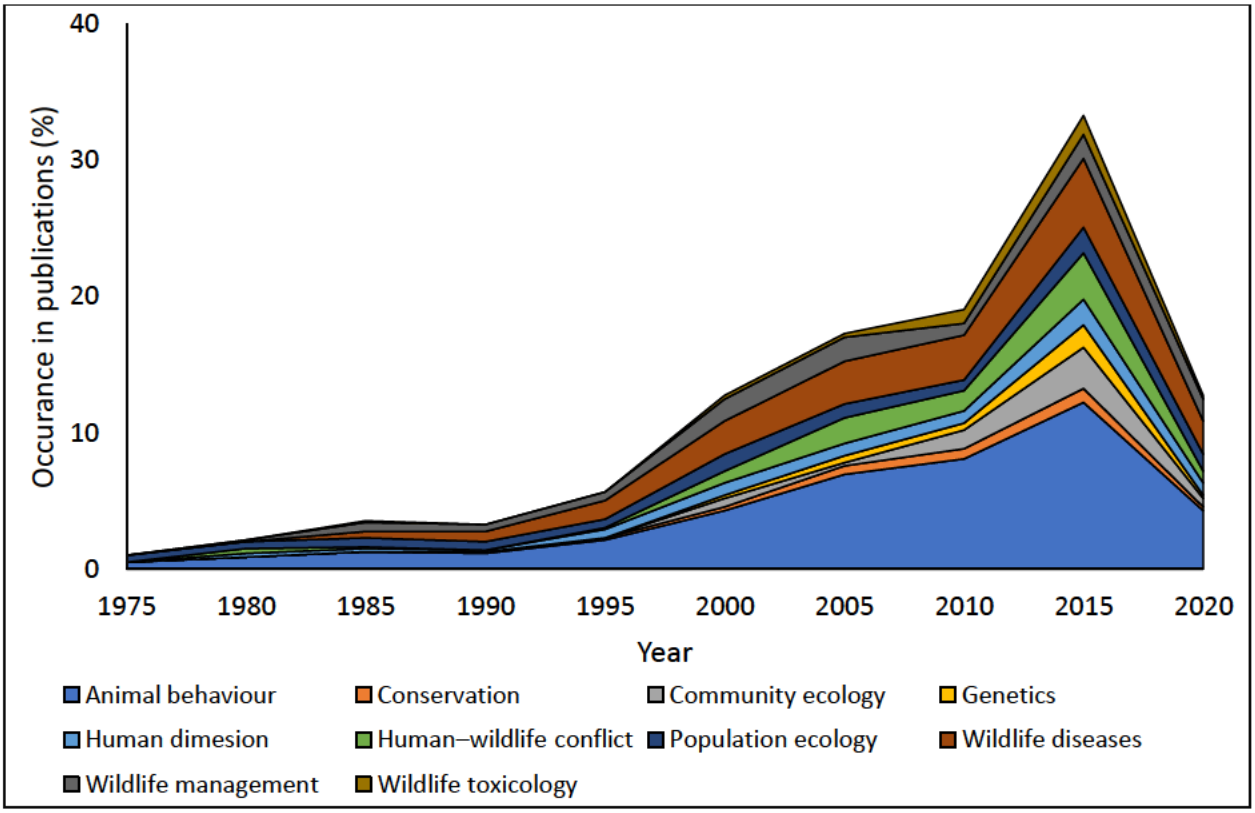


**Figure 2.5.** The percentage occurrence of scientific topics focused on urban mammalian mesocarnivore that have been published in peer-reviewed journal papers.

## 2.5 Discussion

The number of publications and studies focusing on the urban landscape reflects a rapid growth and urgency in researching urban ecology (Adams 2005; Magle et al. 2012; Marzluff et al. 2008). Publications focusing on urban mammalian carnivores have increased two-fold in the last decade compared with the cumulative available literature before 2010. Magle et al. (2012) study demonstrated that there had been a rapid expansion of ecological studies focused on urban fauna. This burgeoning research attention in the urban environment suggests a necessity to fill a novel and expanding knowledge gap in wildlife research (Magle et al. 2012; Marzluff et al. 2008). There is also an urgency is needing to comprehensibly understand the dynamics of an anthropogenic environment in light of the continued decrease in natural habitat available for wildlife (Goddard et al. 2010; McKinney 2006; Saura et al. 2018). Another explanation is simply the global increase of research effort resulting in publications due to the "publish or perish" pressure principle.

Although urban ecology is expanding its multi-disciplinary footprint, the extent of influence appears to be geographically biased. Approximately 45% of all urban mesocarnivore studies have been conducted in North America, and a further 33% limited to Europe. The extent of this bias is accentuated when one considers the discrepancy between northern vs southern hemisphere research output. Approximately 90% of all studies conducted on urban mesocarnivores have originated from the northern hemisphere. Geographical bias is not exclusive to this study. Comparable trends were established in Magle et al. (2012), which looked at seven broad fauna groups (mammals, birds, fishes, reptiles, amphibians, arthropods, and others) in urban systems. In our study, like Magle et al. (2012), the majority of the studies reviewed were English-based journals and mainly based in America, which we expected would impact our results. More recently, several reviews have further demonstrated this North hemisphere bias (Ritzel & Gallo 2020; Fusco et al. 2021; Luna et al. 2021) However, we have

confidence that our results reflect the focus of urban mesocarnivores studies globally as only peer–reviewed journals were used in our review process. Europe was the first continent to implement urban-based ecological surveys which have allowed the field to broaden our understanding of urban systems which mammalian mesocarnivores are integral to (Adams 2005). In this development, the UK was innovatively progressive, which lay the foundation for the environmental consciousness in European countries. The link between conservation and urbanization in urban areas has yet to scratch the surface in the developing world. The low number of studies published in rapidly urbanising regions of the world (Asia, Africa and South America) is of particular concern for global biodiversity. Studies from these continents tend to be uncommon in urban areas with research focus rather dedicated to pristine natural systems (Magle et al. 2012). These regions are experiencing rapid urbanisation, and population growth which comes at the cost of the natural systems (Haddad et al. 2015; Lepczyk et al. 2017; Marzluff 2001; United Nations 2019). Unfortunately, much of the natural, rich heritage is located in developing countries of the world are located in economically "poor" countries (Ceballos & Ehrlich 2006; Myers et al. 2000). Another explanation for the disparity among studies on mesocarnivores geographically centres around human population densities and nature of the urban area (city centre, town, or suburb), which are suggested to provide more opportunities to generalist medium-sized urban carnivores (Bateman & Fleming 2012; Gehrt et al. 2010). Bateman and Fleming (2012) reported that no urban carnivore studies had been conducted in Africa besides anecdotal information before 2012. They attributed this to the nature of urbanisation and predator guilds on the continent. Our review showed similar findings, and that urban ecology in Africa (South Africa and Botswana) is a recently explored field (Table 1). However, several field studies have highlighted that urbanised landscapes in South Africa are potentially experiencing mesopredator release. Increased sightings of mesocarnivores have been documented in several South African cities, suburbs and towns, and these have

subsequently seen an increase in research in the last five years (Appendix S1) (Alexander et al. 2019b, 2019c; Serieys et al. 2019a; Widdows & Downs 2018; Zungu et al. 2020a; Zungu et al. 2020b). With an emphasis on South Africa, we suggest that the movement of mesocarnivores in urban areas is not a recent event and that they have perhaps been present for many years. This has been evidenced in the form of anecdotal information. Instead of what, researchers have recently acknowledged the gap in urban studies in South Africa. Wildlife researchers are progressively shifting attention to include our anthropogenically modified landscape studies opposed to focusing on natural wildlife systems.

A metropolitan city in South Africa, Durban- or currently referred to as eThekwini Municipality, established a green corridor system in the 1980s, an innovative conservation measure, to preserve the local flora and fauna (Roberts 1994). Known as the Durban Metropolitan Open Space System (D'MOSS), the long-term conservation measure was based on principles of island biogeography (Roberts 1994). Its core natural areas are linked with connecting corridors and buffer areas to conserve the primary function and wildlife in these spaces. Despite this promising start, limited research has been conducted on mammalian carnivores that persist in these green spaces. Seven different species of mesocarnivores presently inhabit these D'MOSS areas on the fringe and within the urban matrix (African wild cat: *Felis lybica caffra*, banded mongoose: *Mungos mungo*, caracal: *Caracal caracal*, cape genet: *Genetta tigrina*, large-grey mongoose: *Herpestes ichneumon*, water mongoose *Atilax paludinosus*, slender mongoose: *Galerella sanguinea* and white-tailed mongoose: *Ichneumia albicauda*) (Widdows et al. 2015; Zungu et al. 2020a). In the last five years, there has been an increase of research effort to increase our understanding of habitat fragmentation and disturbance on fauna in general in KwaZulu-Natal (Alexander et al. 2019b, 2019c; Ehlers Smith et al. 2020; Maseko et al. 2020; McPherson et al. 2019; Widdows & Downs 2018; Zungu et al. 2020a; Zungu et al. 2020b). Similar types of studies have been recently conducted in the City

of Cape Town, Johannesburg and Tshwane (Cronk & Pillay 2018, 2019a, 2019b, 2020; Keyes et al. 2020; Koeppel et al. 2020; Leighton et al. 2020; Okes & O'Riain 2017; Okes & O'Riain 2019; Ponsonby & Schwaibold 2018; Schnetler et al. 2020; Serieys et al. 2019a; Serieys et al. 2019b; Viljoen et al. 2020). Nevertheless, the effects of urbanisation on mammalian carnivores currently remain inadequately known for South Africa despite the recent expansion of the field. More concerningly is that 19 studies conducted in South Africa constituted 90% of all urban research for the entire continent.

We found 61 species represented in our global literature analysis of urban mammalian mesocarnivores. Of these 61 species, only two appear to be meaningfully studied within urban literature (red fox and coyote). Despite the adverse effects associated with urbanisation, several species (red fox, coyotes, raccoon and stone marten) are suggested to be synurbic, that is, thriving in the urban landscapes (Bateman & Fleming 2012; Šálek et al. 2015). North hemispheric mesocarnivores represented the most researched species in urban environments (except for the red fox). The southern hemisphere does not yet boast a single research species, which fits the synurbic model to date. The lag in research on mesocarnivores in urban spaces in developing countries represents the wildlife conservation priorities in these regions. Developing counties currently are invested in conserving larger charismatic wildlife in natural landscapes as this is often linked with income earning potential (Lindsey et al. 2017). Certain South American and African countries have been shown to be significant contributors in the global conservation fight, with the hope that an umbrella conserving approach will indirectly benefit a suite of other species (Lindsey et al. 2017; Macdonald et al. 2012).

Canids were the most frequently studied family in our review. Canids are represented by some of the most adaptable wild carnivore species. Their extreme adaptability has allowed taxa of the family to inhabit the most expansive geographical ranges spanning diverse habitats (arctic tundra, arid deserts and city centres). The red fox epitomises this. Their behavioural

plasticity, social ecology and dietary habits, in the absence of human persecution, has allowed medium-sized red foxes and cayotes to exploit urban environments. Red foxes and coyotes represented the most broadly studied species in our review. Previously thought as a British city phenomenon, red foxes have truly established as metropolitan species, residing in 114 cities across the global (Gehrt et al. 2010). The behavioural and morphological characteristics have predisposed red foxes to be successful colonisers, particularly when introduced to countries in the absence of competition (Marks & Bloomfield 1999; Scott et al. 2014; Wilkinson & Smith 2001). Coyotes also appear to own the dexterity and suite of traits that allow them to be a synurbic species (Gehrt et al. 2010; Murray & St Clair 2015). Despite an increase in population numbers being documented in the coyote populations, to and thrive in most environments modified by humans in northern America (Gehrt et al. 2011), limited information on their ecology is presently known in comparison to rural populations. Other candid species appear to be less voracious exploiters of anthropogenically modified landscapes. “Urban dwellers” such as the grey fox (Kapfer & Kirk 2012) and kit fox (Harrison et al. 2011) appear to be poorer adapters of urban landscapes and only occurring on urban margins (sub-urban and exurban area). “Urban dwellers” are subsequently less researched. In the same way, species belonging to other families of Carnivora have not been studied to the same depth as established synurbic Canidae species (Table 1; Appendix S1).

Human–wildlife interactions (HWI), especially with mesocarnivores, appear to have initiated the pursuit of urban studies of mammalian mesocarnivores in developed countries. Nuisance coyotes of the Chicago metropolitan area experience anthropogenic control measures because of population expansion (Gehrt et al. 2011). Similarly, in Japan, raccoons are well-acquainted with city-life since their introduction to the island and are now regarded as a pest species (Ikeda et al. 2004). Defensive behaviour by striped skunks elicits a traumatic response for humans and domestic pets (Wood 1999). Consequently, information on population control

practises having been intently pursued (Prange & Gehrt 2004). Dumpster diving habits and property damage by urban badgers have seen an increase in conservation efforts to understand the species ecology in urban environments (Davison et al. 2008). HWI appears to be the driving force behind investigations on urban mesocarnivore populations and behavioural activity to address these interactions and mitigate harmful/adverse conflicts. HWI in developing countries are largely still considered rural-based (Messmer 2000), and for this reason, urban studies on mesocarnivores appear to be underrepresented because resources are directed to rural areas. Furthermore, the field is still relatively young in developed countries (Magle et al. 2012). It is expected that the shift in research-focus to urban areas in developing countries will be slow.

The study of urban mammalian mesocarnivores is a rapidly emerging field with interest in a broad range of disciplines. Animal behaviour remains the popular topic of urban mesocarnivore research across the decades with researchers focusing studies on how species spatial interpret and react to anthropogenically modified environments (Magle et al. 2012). Wildlife disease was the next common field of interest. There has been a major drive to improve understanding of zoonotic disease and parasites on wildlife and human, as well as predict the cycles and impacts of wildlife diseases in the urban environment (Bradley & Altizer 2007). With the diminishing proximity between humans and wildlife, the field is predicted to continue to expand. Besides animal behaviour and wildlife diseases, urban mesocarnivores studies are rarely devoted to a single focal discipline. Research encompassing a diverse range of fields, which allow researchers and conservationists to interpret the impacts more effectively of urbanisation on mesocarnivores, is becoming more common. Wildlife toxicology and genetic studies appear to be on the rise. A lack of connectivity and habitat fragmentation are adversely affecting the genetic health of urban mammals (population sizes and genetic diversity) (Schmidt et al. 2020). Furthermore, non-target secondary poisoning (Anticoagulant rodenticides) and

toxin-based studies (particularly Polychlorinated Biphenyl) associated with human industry and the bioaccumulation of heavy metals are increasingly studied (Serieys et al. 2019a).

Urban wildlife ecology is a prominent field of science that we expect will continue to mushroom as long as urbanization places humans and nature in closer contact. Our review of urban mammalian mesocarnivores underscores the progressive increase of publications that centred around such behaviourally adaptive species. There persists an inordinate amount of research which has been conducted and published in North America and Europe which excludes urbanising regions of Asia, South America, and Africa. Unfortunately, these areas contain the majority of the worlds remaining biodiversity hotspots (Ripple et al. 2017). The current knowledge gap of the effects of urbanisation and its indirect processes on urban mesocarnivores in rich biodiversity hotspots regions is concerning (Ceballos & Ehrlich 2006; Myers et al. 2000).

Our review further showed that several species in the northern hemisphere were extensively studied when directly compared with their southern counterparts. However, when compared with studies conducted in non-urban areas, urban literature is severely lagging (< 2%) (Adams 2005; Magle et al. 2012). The lack of research is inhibiting global conservation efforts, especially considering the inevitable threats urbanisation poses to biodiversity. The inadequacy of information on urban ecology currently represents a critical knowledge gap. Equally, the knowledge gap represents a substantial opportunity in the future of wildlife research to investigate exciting research questions and problems mesocarnivores face. One cannot ignore the paucity of global information on urban wildlife than studies conducted in natural systems. Although urban ecology is an expanding field of science which will rapidly evolve, we still urge investment into the multi-disciplinary field globally. We encourage researchers to explore the landscape on the periphery of the urban core (sub-urban, peri-urban and exurban). The fringe and margins of urban areas are expected to become essential refuge areas for ill-adapted urban species (Parsons et al. 2018).

Our understanding of urban mammalian mesocarnivores ability to adapt and persist in anthropogenically dominated landscapes will evolve as research output is increased, and technology advances. However, presently our current knowledge points out critical distributional and family knowledge gaps which require addressing. Urban wildlife microcosms represent a unique research opportunity with untapped potential for interdisciplinary partnerships in conserving and protecting the wildlife of our planet.

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## 2.8 Supplementary

**Supplementary information Table S2.1.** The total number of studies focused on urban mammalian mesocarnivores that have been published in peer-reviewed journal papers. Publications are categorised by global region and country of origin which the study was conducted.

| Region | Country | Common name | Scientific name | Number of studies | Reference |
|---|---|---|---|---|---|
| Africa | Botswana | Banded mongoose | *Mungos mungo* | 2 | (Flint et al. 2016; Medley et al. 2020) |
| | South Africa | Black-backed jackal | *Canis mesomelas* | 2 | (Keyes et al. 2020; Koeppel et al. 2020) |
| | | Cape clawless otter | *Aonyx capensis* | 5 | (Okes & O'Riain 2017; Okes & O'Riain 2019; Ponsonby & Schwaibold 2018; Schnetler et al. 2020; Serieys et al. 2019a) |
| | | Caracal | *Caracal caracal* | 5 | (Leighton et al. 2020; Schnetler et al. 2020; Serieys et al. 2019a; Serieys et al. 2019b; Viljoen et al. 2020) |
| | | Cape genet | *Genetta tigrina* | 6 | (Schnetler et al. 2020; Widdows & Downs 2015, 2016, 2018; Widdows et al. 2015; Zungu et al. 2020) |
| | | Yellow Mongoose | *Cynictis penicillata* | 5 | (Cronk & Pillay 2018, 2019a, 2019b, 2020; Keyes et al. 2020) |
| | | Small grey mongoose | *Galerella pulverulenta* | 1 | (Schnetler et al. 2020) |
| | | Large grey mongoose | *Herpestes ichneumon* | 1 | (Schnetler et al. 2020) |
| | | Honey badger | *Mellivora capensis* | 2 | (Keyes et al. 2020; Schnetler et al. 2020) |
| | | Striped polecat | *Ictonyx striatus* | 1 | (Schnetler et al. 2020) |
| | | Common genet | *Genetta genetta* | 1 | (Schnetler et al. 2020) |
| | | Cape fox | *Vulpes chama* | 1 | (Schnetler et al. 2020) |

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| | | Water mongoose | *Atilax paludinosus* | 2 | (Keyes et al. 2020; Schnetler et al. 2020) |
| | | Slender mongoose | *Galerella sanguinea* | 4 | (Cronk & Pillay 2018, 2019a; Keyes et al. 2020) |
| | | Spotted-necked otter | *Hydrictis maculicollis* | 1 | (Ponsonby & Schwaibold 2018) |
| Europe | Albania & Macedonia | Eurasian lynx | *Lynx lynx* | 1 | (Trajçe et al. 2019) |
| | Austria | Red fox | *Vulpes vulpes* | 3 | (Duscher et al. 2005; Walter et al. 2018; Wanha et al. 2005) |
| | Belgium | Red fox | *Vulpes vulpes* | 1 | (Brochier et al. 2007) |
| | Bulgaria | European stone marten | *Martes foina* | 1 | (Hisano et al. 2016) |
| | Croatia | European stone marten | *Martes foina* | 1 | (Bilandzic et al. 2010) |
| | | Red fox | *Vulpes vulpes* | 2 | (Bilandzic et al. 2010; Lojkic et al. 2016) |
| | Czech Republic | Ermine stoat | *Mustela erminea* | 1 | (Červinka et al. 2014) |
| | | Least weasel | *Mustela nivalis* | 1 | (Červinka et al. 2014) |
| | | Red fox | *Vulpes vulpes* | 2 | (Červinka et al. 2014; Jankovska et al. 2016) |
| | | Stone marten | *Martes foina* | 2 | (Červinka et al. 2014; Holisova & Obrtel 1982) |
| | Denmark | European badger | *Meles meles* | 1 | (Aaris-Sørensen 1987) |
| | | Stone marten | *Martes foina* | 1 | (Elmeros et al. 2018) |
| | | European polecat | *Mustela putorius* | 1 | (Elmeros et al. 2018) |
| | | Red fox | *Vulpes vulpes* | 1 | (Willingham et al. 1996) |
| | Estonia | Red fox | *Vulpes vulpes* | 2 | (Laurimaa et al. 2015; Plumer et al. 2014) |
| | Finland | Red fox | *Vulpes vulpes* | 2 | (Kauhala et al. 2016; Vuorisalo et al. 2014) |
| | | Raccoon dog | *Nyctereutes procyonoides* | 1 | (Kauhala et al. 2016) |
| | | European badger | *Meles meles* | 1 | (Kauhala et al. 2016) |
| | France | Red fox | *Vulpes vulpes* | 4 | (Castañeda et al. 2020; Comte et al. 2013; Robardet et al. 2011; Robardet et al. 2008) |

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| | | European badger | *Meles meles* | 1 | (Tarabon et al. 2019) |
| | Germany | Red fox | *Vulpes vulpes* | 12 | (Frolich et al. 2000; Frolich et al. 2005; Geduhn et al. 2015; Gras et al. 2018; Immelt et al. 2009; Janko et al. 2012; König 2007; König et al. 2012; Konig et al. 2019; Lempp et al. 2017; Scholz et al. 2020; Schuster et al. 2001) |
| | | Stone marten | *Martes foina* | 3 | (Frolich et al. 2000; Frolich et al. 2005; Lempp et al. 2017) |
| | | Raccoon dog | *Nyctereutes procyonoides* | 3 | (Frolich et al. 2000; Frolich et al. 2005; Lempp et al. 2017) |
| | | European pine marten | *Martes martes* | 2 | (Frolich et al. 2000; Frolich et al. 2005) |
| | | Raccoon | *Procyon lotor* | 3 | (Hohmann et al. 2001; Renteria-Solis et al. 2014a; Renteria-Solis et al. 2014b) |
| | Hungary | European stone marten | *Martes martes* | 2 | (Apathy 1998; Szekeres et al. 2019; Tóth et al. 2009) |
| | | Least weasel | *Mustela nivalis* | 1 | (Szekeres et al. 2019) |
| | Italy | Stone marten | *Martes foina* | 2 | (Lucherini & Crema 1993; Prigioni et al. 2008) |
| | | Eurasian Otter | *Lutra lutra* | 1 | (Marcelli & Fusillo 2009) |
| | | European badger | *Meles meles* | 2 | (Balestrieri et al. 2009; Prigioni et al. 2008) |
| | | Common genet | *Genetta genetta* | 1 | (Molina-Vacas et al. 2012) |
| | | Red fox | *Vulpes vulpes* | 2 | (Molina-Vacas et al. 2012; Prigioni et al. 2008) |
| | | European pine marten | *Martes martes* | 1 | (Lombardini et al. 2015; Prigioni et al. 2008) |
| | Luxembourg | European stone marten | *Martes foina* | 3 | (Herr et al. 2010; Herr et al. 2009a, 2009b) |
| | Netherlands | European stone marten | *Martes foina* | 1 | (Piza-Roca et al. 2018) |
| | Norway | European badger | *Meles meles* | 3 | (Bevanger et al. 1996; Bjerke & Ostdahl 2004; Gomes et al. 2020) |
| | | Eurasian lynx | *Lynx lynx* | 3 | (Bouyer et al. 2015; Kleiven et al. 2004; Strand et al. 2019) |
| | | Red fox | *Vulpes vulpes* | 2 | (Bjerke & Ostdahl 2004; Mo et al. 2018) |
| | | Wolverine | *Gulo gulo* | 2 | (Kleiven et al. 2004; Strand et al. 2019) |

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| | Poland | Red fox | *Vulpes vulpes* | 4 | (Dobrzański et al. 2014; Duduś et al. 2014; Jakubas et al. 2018; Tomza-Marciniak et al. 2012) |
| | | Least weasel | *Mustela nivalis* | 1 | (Łopucki et al. 2019) |
| | | Raccoon dog | *Nyctereutes procyonoides* | 1 | (Jakubas et al. 2018) |
| | | Eurasian otter | *Lutra lutra* | 1 | (Jakubas et al. 2018) |
| | | Raccoon | *Procyon lotor* | 1 | (Bartoszewicz et al. 2008) |
| | | European badger | *Meles meles* | 1 | (Jakubas et al. 2018) |
| | | Stone marten | *Martes foina* | 2 | (Duduś et al. 2014; Wereszczuk et al. 2017) |
| | | Arctic fox | *Vulpes lagopus* | 1 | (Dobrzański et al. 2014) |
| | Portugal | European stone marten | *Martes foina* | 1 | (Grilo et al. 2009) |
| | | Red fox | *Vulpes vulpes* | 2 | (Alexandre et al. 2020; Grilo et al. 2009) |
| | | European polecat | *Mustela putorius* | 1 | (Grilo et al. 2009) |
| | | Least weasel | *Mustela nivalis* | 1 | (Grilo et al. 2009) |
| | | Eurasian Otter | *Lutra lutra* | 1 | (Grilo et al. 2009) |
| | | Common genet | *Genetta genetta* | 1 | (Grilo et al. 2009) |
| | | European badger | *Meles meles* | 1 | (Grilo et al. 2009) |
| | | Large grey mongoose | *Herpestes ichneumon* | 1 | (Grilo et al. 2009) |
| | Slovak Republic | Red fox | *Vulpes vulpes* | 1 | (Antolova et al. 2004) |
| | Spain | American mink | *Neovison vison* | 2 | (Delibes et al. 2004; Lopez-Perea et al. 2019) |
| | | European badger | *Meles meles* | 3 | (Fandos et al. 2012; Lopez-Perea et al. 2019; Millán et al. 2016) |
| | | European polecat | *Mustela putorius* | 2 | (Lopez-Perea et al. 2019; Zabala et al. 2005) |

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| | | European stone marten | *Martes foina* | 5 | (Fandos et al. 2012; Lopez-Perea et al. 2019; Millán et al. 2016; Recio et al. 2015; Vergara et al. 2015) |
| | | European pine marten | *Martes martes* | 2 | (Lopez-Perea et al. 2019; Vergara et al. 2015) |
| | | Red fox | *Vulpes vulpes* | 4 | (Fandos et al. 2012; Lopez-Perea et al. 2019; Millán et al. 2016; Recio et al. 2015) |
| | | Wild cat | *Felis silvestris* | 2 | (Lopez-Perea et al. 2019; Recio et al. 2015) |
| | | Eurasian Otter | *Lutra lutra* | 2 | (Lopez-Perea et al. 2019; Millán et al. 2016) |
| | | Common genet | *Genetta genetta* | 3 | (Fandos et al. 2012; Lopez-Perea et al. 2019; Millán et al. 2016) |
| | Sweden | European polecat | *Mustela putorius* | 1 | (Karlsson & Sjostrom 2008) |
| | | Wolverine | *Gulo gulo* | 1 | (Ericsson et al. 2007) |
| | | Eurasian lynx | *Lynx lynx* | 1 | (Karlsson & Sjostrom 2008) |
| | Switzerland | Eurasian Otter | *Lutra lutra* | 1 | (Weber 2011) |
| | | European badger | *Meles meles* | 1 | (Geiger et al. 2018) |
| | | European pine marten | *Martes martes* | 1 | (Weber et al. 2018) |
| | | European stone marten | *Martes foina* | 1 | (Kistler et al. 2013) |
| | | Red fox | *Vulpes vulpes* | 17 | (Bontadina et al. 2001; Contesse et al. 2004; DeCandia et al. 2019; Deplazes et al. 2004; Dip et al. 2003; Dip et al. 2001; Fischer et al. 2005; Geiger et al. 2018; Gloor et al. 2001; Hegglin et al. 2007; Hegglin et al. 2004; Hegglin & Deplazes 2008; Hegglin et al. 2003; Hofer et al. 2000; Reperant et al. 2007; Stieger et al. 2002; Wandeler et al. 2003) |
| | UK | European badger | *Meles meles* | 13 | (Baker & Harris 2007; Clarke et al. 1998; Cresswell & Harris 1988a, 1988b; Davison et al. 2008, 2009; Davison et al. 2011; Delahay et al. 2009; Harris 1984; Huck et al. 2008a; Huck et al. 2008b; Scott et al. 2018; Ward et al. 2016) |
| | | Red fox | *Vulpes vulpes* | 53 | (Arnold et al. 2011; Atterby et al. 2015; Baker et al. 2007; Baker et al. 2004a; Baker et al. 2004b; Baker et al. 2000; Baker & Harris 2000; Baker & Harris 2007; Baker et al. 1998; Blanch-Lázaro et al. 2018; Doncaster & Macdonald 1997; Doncaster et al. 1990; Doncaster & Macdonald 1991; Doncaster & Macdonald 1992; Dorning & Harris 2017, |

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| | | | | | 2019a, 2019b, 2019c; Fawcett et al. 2017; Harris 1977, 1979, 1981a, 1981b; Harris & Rayner 1986a; Harris & Rayner 1986b, 1986c; Harris & Smith 1987; Harris et al. 1988; Harris & Trewhella 1988; Inger et al. 2016; Iossa et al. 2009; Kolb 1984, 1985, 1986; Newman et al. 2003; Page 1981; Potts et al. 2013; Richards et al. 1993, 1995; Robertson et al. 2000; Saunders et al. 1997; Scott et al. 2018; Scott et al. 2020; Scott et al. 2014; Smith & Harris 1991; Smith & Woods 2007; Soulsbury et al. 2008; Tolhurst et al. 2016; Trewhella & Harris 1988, 1990; White & Harris 1994; White et al. 1996; Whiteside et al. 2011; Wilkinson & Smith 2001) |
| North America | Canada | North American river otter | *Lontra canadensis* | 3 | (Guertin et al. 2010a; Guertin et al. 2010b; Nelson et al. 2015a) |
| | | Coyote | *Canis latrans* | 8 | (Alexander & Quinn 2011; Cluff 2006; Liccioli et al. 2015a; Liccioli et al. 2012a; Liccioli et al. 2012b; Liccioli et al. 2014; Luong et al. 2018; Murray & St Clair 2017) |
| | | Raccoon | *Procyon lotor* | 4 | (Broadfoot et al. 2001; Jardine et al. 2011; Rosatte et al. 1992; Schubert et al. 1998) |
| | | Red fox | *Vulpes vulpes* | 5 | (Adkins & Stott 1998; Rosatte & Allan 2009; Rosatte et al. 1993; Rosatte et al. 2007; Rosatte et al. 1992) |
| | | Striped skunk | *Mephitis mephitis* | 5 | (Britton et al. 2017; Britton et al. 2019; Broadfoot et al. 2001; Rosatte et al. 2011; Rosatte et al. 1992) |
| | United states | American Badger | *Taxidea taxus* | 3 | (Crooks 2002; Ordeñana et al. 2010; Quinn et al. 2012) |
| | | American Mink | *Neovison vison* | 3 | (Ahlers et al. 2016; Mech 2003; Moll et al. 2020a) |
| | | American river otter | *Lontra canadensis* | 2 | (Hanrahan et al. 2019; Mech 2003) |
| | | Bobcat | *Lynx rufus* | 48 | (Alonso et al. 2015; Carver et al. 2016; Chupp et al. 2013; Crooks 2002; Dunagan et al. 2019; Eakin et al. 2018; Fedriani et al. 2000; Flores-Morales et al. 2019; Fountain-Jones et al. 2017; George & Crooks 2006; Goad et al. 2014; Harrison 1998; Kowalski et al. 2015; Kozakiewicz et al. 2019; Larson et al. 2015; Lewis et al. 2015a; Lewis et al. 2015b; Lewis et al. 2017a; Lewis et al. 2017b; Lombardi et al. 2017; Markovchick-Nicholls et al. 2008; Morzillo & Schwartz 2011; Nickel et al. 2020; Ordeñana et al. 2010; Parsons et al. 2019; Patten & Burger 2018; Poessel et al. 2014a; Reed et al. 2017; Riley et al. 2004; Riley et al. 2006; Riley 2006; Riley et al. 2007; Riley et al. 2003; Ruell et al. 2012; Ruell et al. 2009; Serieys et al. 2015a; Serieys et al. 2015b; Smith et al. 2018; Smith et al. 2020; |

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| | | | | | Stark et al. 2020; Steinberg et al. 2015; Tigas et al. 2002; Tigas et al. 2003; Tracey et al. 2013; Wait et al. 2018; Windell et al. 2020; Young et al. 2019a; Young et al. 2019b) |
| | | Coyote | *Canis latrans* | 116 | (Aher et al. 2016; Andelt & Mahan 1980; Atkinson & Shackleton 1991; Atwood 2006; Atwood & Weeks 2003; Atwood et al. 2004; Blue & Alexander 2015; Bombieri et al. 2018; Bounds & Shaw 1997; Breck et al. 2017; Breck et al. 2019; Catalano et al. 2012; Crooks 2002; DeStefano & DeGraaf 2003; Dodge & Kashian 2013; Drake et al. 2020; Drake et al. 2019; Eakin et al. 2018; Ellington & Gehrt 2019; Elliot et al. 2016; Fedriani et al. 2001; Fedriani et al. 2000; Flores-Morales et al. 2019; Franckowiak et al. 2019; Gallo et al. 2019; Gallo et al. 2017; Garwood et al. 2015; Gehrt 2004; Gehrt et al. 2009; Gehrt et al. 2011; Gehrt et al. 2013; George & Crooks 2006; Gese et al. 2012; Gibeau 1998; Goad et al. 2014; Gompper 2002; Gosselink et al. 2003; Greenspan et al. 2018; Grigione et al. 2011; Grigione et al. 2014; Grinder & Krausman 2001a; Grinder & Krausman 2001b; Grubbs & Krausman 2009; Henger et al. 2020; Hennessy et al. 2012; Jackman & Rutberg 2015; Jones et al. 2016; Jordan & Lobb-Rabe 2015; Kays et al. 2015; Kellert 1985; Kenaga et al. 2013; Kowalski et al. 2015; Krausman et al. 2006; Larson et al. 2020; Larson et al. 2015; Lawrence & Krausman 2011; LeFlore et al. 2019; Lewis et al. 2017a; Liccioli et al. 2015b; Lombardi et al. 2017; Lu et al. 2016; Magle et al. 2016; Magle et al. 2014a; Magle et al. 2014b; Malmlov et al. 2014; Markovchick-Nicholls et al. 2008; Mitchell et al. 2015; Moll et al. 2018; Moll et al. 2020a; Moll et al. 2020b; Morey et al. 2007; Morzillo & Schwartz 2011; Mueller et al. 2018, 2019; Murray et al. 2015a; Murray et al. 2015b; Murray et al. 2016; Murray & St Clair 2015; Newsome et al. 2015; Nickel et al. 2020; Ordeñana et al. 2010; Parsons et al. 2019; Patten & Burger 2018; Poessel et al. 2015; Poessel et al. 2016; Poessel et al. 2013; Poessel et al. 2014b, 2017a; Poessel et al. 2017b; Quinn 1995, 1997a, 1997b; Randa & Yunger 2006; Riley et al. 2006; Riley et al. 2003; Santana & Armstrong 2017; Smith et al. 2018; Sponarski et al. 2016; Stark et al. 2020; Stevens et al. 1994; Tigas et al. 2002; Tigas et al. 2003; Wait et al. 2018; Watts et al. 2015; Way et al. 2001; Way et al. 2006; Way & Eatough 2006; Way et al. 2004; Weckel & Wincorn 2016; Weckel et al. 2010; Wilson & Rose 2019; Windell et al. 2020; Wine et al. 2015; Wurth et al. 2020; Young et al. 2019c) |
| | | Fisher | *Pekania pennanti* | 2 | (Eakin et al. 2018; Gabriel et al. 2012) |
| | | Grey fox | *Urocyon cinereoargenteus* | 20 | (Chupp et al. 2013; Crooks 2002; Eakin et al. 2018; Fedriani et al. 2000; Gabriel et al. 2009a; Gabriel et al. 2009b; Kapfer & Kirk 2012; Kays & Parsons 2014; Kowalski et al. 2015; Larson et al. 2015; LeFlore et al. 2019; Lombardi et al. 2017; Markovchick- |

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| | | | | | Nicholls et al. 2008; Ordeñana et al. 2010; Parsons et al. 2019; Patten & Burger 2018; Riley et al. 2004; Riley 2006; Smith et al. 2018; Windell et al. 2020) |
| | | Kit fox | *Vulpes macrotis* | 13 | (Cypher & Warrick 1993; Cypher & Frost 1999; Cypher et al. 2017; Harrison et al. 2011; Montecino-Latorre et al. 2019; Morzillo & Schwartz 2011; Nelson et al. 2015b; Newsome et al. 2010; Nogeire-McRae et al. 2019; Ralls et al. 2007; Riner et al. 2018; Westall et al. 2019a; Westall et al. 2019b) |
| | | Virginia opossum | *Virginia opossum* | 21 | (Brown et al. 2010; Chupp et al. 2013; Crooks 2002; DeStefano & DeGraaf 2003; Greenspan et al. 2018; Haverland & Veech 2017; Jordan & Lobb-Rabe 2015; Kanda et al. 2005; Kays & Parsons 2014; Magle et al. 2016; Magle et al. 2014b; Markovchick-Nicholls et al. 2008; Meckstroth & Miles 2005; Moll et al. 2020a; Moll et al. 2020b; Morzillo & Schwartz 2011; Nickel et al. 2020; Ordeñana et al. 2010; Prange & Gehrt 2004; Stark et al. 2020; Tigas et al. 2003) |
| | | Raccoon | *Procyon lotor* | 39 | (Andelt & Woolley 1996; Anthony et al. 1990; Bozek et al. 2007; Brown et al. 2010; Chupp et al. 2013; Crooks 2002; Demeny et al. 2019; DeStefano & DeGraaf 2003; Eakin et al. 2018; Gehrt 2004; Greenspan et al. 2018; Gross et al. 2012; Haverland & Veech 2017; Hoffmann 1979; Hoffmann & Gottschang 1977; Jordan & Lobb-Rabe 2015; Kays & Parsons 2014; Kowalski et al. 2015; Magle et al. 2016; Magle et al. 2014b; Markovchick-Nicholls et al. 2008; Moll et al. 2020a; Moll et al. 2020b; Morzillo & Schwartz 2011; Ordeñana et al. 2010; Page et al. 2008; Patten & Burger 2018; Prange & Gehrt 2004; Prange et al. 2011; Prange et al. 2003, 2004; Rainwater et al. 2017; Ram et al. 2007; Randa & Yunger 2006; Riley et al. 1998; Stark et al. 2020; Theimer et al. 2015; Tigas et al. 2003; Valentine et al. 1988) |
| | | Red fox | *Vulpes vulpes* | 23 | (DeStefano & DeGraaf 2003; Eakin et al. 2018; Goad et al. 2014; Gosselink et al. 2010; Gosselink et al. 2003; Gosselink et al. 2007; Handler et al. 2020; Kasprowicz et al. 2016; Kimmig et al. 2020; Lavin et al. 2003; LeFlore et al. 2019; Lewis et al. 1999; Lombardi et al. 2017; Moll et al. 2018; Moll et al. 2020a; Moll et al. 2020b; Mueller et al. 2018; Parsons et al. 2019; Randa & Yunger 2006; Sacks et al. 2016; Savory et al. 2014; Stark et al. 2020; Windell et al. 2020) |
| | | San Clemente Island fox | *Urocyon littoralis* | 2 | (Gould & Andelt 2011, 2013) |
| | | Striped skunk | *Mephitis mephitis* | 26 | (Andelt & Woolley 1996; Brashear et al. 2015; Chupp et al. 2013; Crooks 2002; Eakin et al. 2018; Gehrt 2004; Gehrt 2005; Greenspan et al. 2018; Hamilton 1936; Kowalski et al. 2015; Meckstroth & Miles 2005; Moll et al. 2020a; Moll et al. 2020b; Morzillo & Schwartz 2011; Nickel et al. 2020; Oertli et al. 2009; Ordeñana et al. 2010; Patten & |

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| | | | | | Burger 2018; Prange & Gehrt 2004; Schneider et al. 2019; Stark et al. 2020; Theimer et al. 2015; Theimer et al. 2016; Tigas et al. 2003; Wait et al. 2018; Weissinger et al. 2009) |
| | | Long-tailed weasel | *Mustela frenata* | 3 | (Crooks 2002; Moll et al. 2020a; Ordeñana et al. 2010) |
| | | Arctic fox | *Vulpes lagopus* | 1 | (Savory et al. 2014) |
| | | Western spotted skunk | *Spilogale gracilis* | 2 | (Crooks 2002; Ordeñana et al. 2010) |
| Asia & Pacific | Australia | Dingo | *Canis lupus dingo* | 4 | (Allen et al. 2016; Allen et al. 2013; McNeill et al. 2016; Wysong et al. 2020) |
| | | Red fox | *Vulpes vulpes* | 14 | (Coman et al. 1991; Hradsky et al. 2017; Huijbers et al. 2013; Jackson et al. 2007; Jenkins & Craig 1992; Loh et al. 2018; Marks & Bloomfield 1998; Marks & Bloomfield 1999a, 1999b, 2006; Marks et al. 1996; Robinson & Marks 2001; Villaseñor et al. 2015; White et al. 2006) |
| | Japan | European badger | *Meles meles* | 1 | (Kaneko et al. 2006) |
| | | Japanese badger | *Meles meles anakuma* | 3 | (Kaneko 2005; Tanaka 2005; Tanaka et al. 2002) |
| | | Japanese weasel | *Mustela itatsi* | 3 | (Kameo et al. 2012; Kaneko et al. 2009; Okawara et al. 2014) |
| | | Raccoon | *Procyon lotor* | 4 | (Ikeda et al. 2004; Lee et al. 2011; Saito & Koike 2013; Saito & Koike 2015) |
| | | Raccoon dog | *Nyctereutes procyonoides* | 13 | (Abe et al. 2010; Enomoto et al. 2018; Goto et al. 1999; Kameo et al. 2012; Mitsuhashi et al. 2018; Nummi et al. 2019; Saito & Koike 2013; Saito & Koike 2015; Saito & Sonoda 2017; Saito et al. 2016; Soga & Koike 2013; Takeuchi et al. 2012; Tsunoda et al. 2019) |
| | | Red fox | *Vulpes vulpes* | 7 | (Amaike et al. 2018; Ikeda et al. 2014; Inoue et al. 2007; Kato et al. 2017; Nonaka et al. 2006; Tsukada et al. 2000; Uraguchi et al. 2009) |
| | | Masked civet | *Paguma larvata* | 3 | (Abe et al. 2010; Lee et al. 2011; Saito & Koike 2015) |
| | South Korea | Eurasian otter | *Lutra lutra* | 1 | (Hong et al. 2017) |
| | | Raccoon dog | *Nyctereutes procyonoides* | 1 | (Lee & Rhim 2017) |
| | | European badger | *Meles meles* | 1 | (Lee & Rhim 2017) |
| | | Siberian weasel | *Mustela sibirica* | 1 | (Lee & Rhim 2017) |

| | | | | | |
|---|---|---|---|---|---|
| | | Yellow-throated marten | *Martes flavigula* | 1 | (Lee & Rhim 2017) |
| | Thailand | Smooth-coated otter | *Lutrogale perspicillata* | 1 | (Kamjing et al. 2017) |
| | Taiwan | Clouded leopard | *Neofelis nebulosa* | 1 | (Greenspan et al. 2020) |
| | | Ferret badger | *Melogale moschata* | 2 | (Liu et al. 2020; Yen et al. 2019) |
| | | Masked palm civet | *Paguma larvata* | 2 | (Liu et al. 2020; Yen et al. 2019) |
| | | Crab-eating mongoose | *Herpestes urva* | 1 | (Liu et al. 2020) |
| | | Small Indian civet | *Viverricula indica* | 1 | (Yen et al. 2019) |
| South America | Brazil | South American coati | *Nasua nasua* | 9 | (Almeida et al. 2020; Alves-Costa et al. 2004; Costa et al. 2009; Costa et al. 2018; Ferreira et al. 2013; Ferro et al. 2019; Garcia et al. 2013; Rodrigues et al. 2006; Silva et al. 2014) |
| | | Crab-eating fox | *Cerdocyon thous* | 3 | (Facure & MonteiroFilho 1996; Garcia et al. 2013; Pedó et al. 2006) |
| | | Maned wolf | *Chrysocyon brachyurus* | 2 | (Lion et al. 2011; Massara et al. 2015) |
| | Chile | American mink | *Neovison vison* | 2 | (Barros et al. 2018; Sepulveda et al. 2011) |
| | | Southern river otters | *Lontra provocax* | 1 | (Barros et al. 2018) |
| | Colombia | Crab-eating fox | *Cerdocyon thous* | 1 | (Bedoya et al. 2018) |
| | | Long-tailed weasel | *Mustela frenata* | 1 | (Bedoya et al. 2018) |
| | Costa Rica | Raccoon | *Procyon lotor* | 1 | (Baldi et al. 2019) |
| | Mexico | Ring-tailed cat | *Bassariscus astutus* | 4 | (Barja & List 2006; Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b; Ramírez-Cruz 2019) |

| | | | | | |
|---|---|---|---|---|---|
| | | White-nosed coati | *Nasua narica* | 2 | (de-la-Rosa-Arana et al. 2016) |
| | | Raccoon | *Procyon lotor* | 3 | (de-la-Rosa-Arana et al. 2016; Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b) |
| | | Common opossum | *Didelphis marsupialis* | 2 | (Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b) |
| | | Grey four-eyed opossum | *Philander opossum* | 2 | (Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b) |
| | | Long-tailed weasel | *Mustela frenata* | 2 | (Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b) |
| | | Grey fox | *Urocyon cinereoargenteus* | 2 | (Mella-Méndez et al. 2019a; Mella-Méndez et al. 2019b) |

## CHAPTER 3

# Home range and core area utilisation of three co-existing mongoose species: large grey, water and white-tailed in the fragmented landscape of the KwaZulu-Natal Midlands, South Africa

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**Running header:** Spatial ecology of three mongoose species

### 3.1 Abstract

The spatial ecology of the Herpestidae family has remained poorly studied across Africa. The behavioural plasticity and generalist nature of members of the family could be facilitating their expansion in anthropogenically transformed landscapes. Given the current paucity of information on their spatial ecology, knowledge of their spatial movement is important ecological information for the species conservation. Three co-existing mongoose species (large grey (*Herpestes ichneumon*, n = 5)), water (*Atilax paludinosus*, n = 5) and white-tailed (*Ichneumia albicauda*, n = 2)) were collared and tracked from September 2016 – October 2017 using Global Positioning System (GPS)-Ultra high frequency (UHF) transmitters to determine their home range size and fine-scale spatial movement in the fragmented natural habitat and farmland mosaic landscape of the KwaZulu-Natal Midlands, South Africa. Three home range methods (minimum convex polygon (MCP), kernel density estimation (KDE) and local convex hull (LoCoH)) were used to delineate individual home range size and core area utilisation. The overall mean home range size (95% KDE mean ± S.E.) differed among species: large grey (9.8 ± 8.19 $km^2$), water (13.7 ± 5.30 $km^2$) and white-tailed mongoose (0.9 ± 0.06 $km^2$). The mean core area utilisation size (50% KDE means ± S.E.) for large grey, water and white-tailed mongoose was 2.2 ± 0.77 $km^2$, 3.1 ± 0.96 $km^2$ and 0.2 ± 0.02 $km^2$. Species-specific variability in home range size of the study species emphasises this family's adaptability to their surrounding environment in a changing natural habitat and farmland mosaic landscape. The reduced core area use possibly indicates the availability of high resourceful areas and adequate resources within a comparably small area.



***Keywords***: GPS telemetry tracking, Home Range, Mongoose Mosaic agriculture, Small carnivores

## 3.2 Introduction

Globally, biodiversity is threatened by anthropogenic land-use changes and development (McKee et al. 2004; Ripple et al. 2014; Tilman et al. 2017). Anthropogenic land conversion has led to extensive habitat transformation and fragmentation of previously continuous landscapes leading to loss of native biodiversity and the concomitant decline in ecosystem functions (Vitousek et al. 1997; Estes et al. 2011; Di Minin et al. 2016). Species mostly avoid human-transformed landscapes because of the negative impacts that anthropogenic modification imposes on their fitness (Bennett and Saunders 2010; Pekin and Pijanowski 2012). However, generalist species which are often characterised by behavioural plasticity and adaptability often persist in anthropogenically modified landscapes (Sih et al. 2011; Lowry et al. 2013a). Mammalian exploiters of anthropogenic land-use change have been recorded to alter their natural behaviour (diet, activities, home range and dispersal) within these areas and even increase their population densities (Lowry et al. 2013a; Widdows and Downs 2015; Wong and Candolin 2015; Ngcobo et al. 2019a, b).

Many of the world's large mammalian carnivore species have declined with anthropogenic development, habitat fragmentation and land-use change (Ceballos and Ehrlich 2002; Ripple et al. 2014; Ripple et al. 2015; Ripple et al. 2016). In Africa, the relatively high diversity of mammalian carnivore species historically and currently faces high levels of persecution because of conflicts with humans (Ceballos and Ehrlich 2006). In South Africa, large carnivores have been extirpated because of conflicts with humans and to a large extent are now confined to refugia in protected areas and private reserves (Woodroffe et al. 2005; Dickman 2010). The removal of apex mammalian predators from complex systems has influenced the dynamics within the food chain (Estes et al. 2011). The removal of apex carnivores within ecosystems has allowed medium-sized carnivores, generally through "mesocarnivore release", to increase their population densities and ranges, particularly within

anthropogenically modified landscapes such as farmlands (Crooks and Soulé 1999; Prugh et al. 2009; Ritchie and Johnson 2009; Elmhagen et al. 2010; Ripple et al. 2014; Jiménez et al. 2019). Furthermore, favourable life-history traits (e.g. size and sex ratio of offspring, the timing of reproduction, age and size at maturity) in conjunction with behavioural plasticity and a generalist nature have enabled their populations to expand (Prugh et al. 2009; Suraci et al. 2016). Consequently, mesocarnivores then dominate food web interactions in the absence of apex predators (Prugh et al. 2009; Roemer et al. 2009; Suraci et al. 2016). Furthermore, favourable life-history traits (e.g. size and sex ratio of offspring, the timing of reproduction, age and size at maturity) in conjunction with behavioural plasticity and a generalist nature, have enabled their populations to expand (Prugh et al. 2009). Consequently, mesocarnivores then dominate food web interactions in the absence of apex predators (Prugh et al. 2009; Roemer et al. 2009).

Burt (1943) first defined home range as "the area traversed by an individual in its normal activities of food gathering, mating, and caring for young". Occasional sallies outside the area, perhaps exploratory, should not be considered as part of the home range." This definition excludes the activity of dispersal through an individual's lifetime. Home range size is attributed to the physiological state (e.g. sex, size, energetic state, age) and behavioural state (e.g. competition and foraging strategies) of an individual and its interactions within its surroundings (Maddock 1988; Nathan et al. 2008; Cagnacci et al. 2010). Gaining insight into home range size, and distribution (spatial ecology) provides vital information for understanding the impact of the surrounding environment on species life history. Information on the ecological requirements of mammalian carnivores supports suitable conservation strategies for managing their populations.

Herpestidae species are characterised by their adaptability and generalist nature and previous studies in Africa have highlighted their ecological tolerance (e.g. large grey

mongoose: *Herpestes ichneumon*, water mongoose: *Atilax paludinosus*, white-tailed mongoose: *Ichneumia albicauda*, slender mongoose: *Galerella sanguinea* and banded mongoose: *Mungos mungo*) and ability to occupy several habitats across a wide distribution range (Maddock 1988; Rowe-Rowe 1992b; Waser et al. 1995; Ray 1997; Gilchrist and Otali 2002; Ziegler et al. 2002; Skinner and Chimimba 2005; Martinoli et al. 2006; Durant et al. 2010; Pettorelli et al. 2010; Graw et al. 2016). Also, Herpestidae species are among the most widely distributed mammalian carnivores within Africa (Skinner and Chimimba 2005). However, they are generally the most poorly studied mammalian carnivore taxa (Rood 1986). In particular, there are relatively few studies on the spatial ecology of mongoose species across Africa (Skinner and Chimimba 2005; Table 3.1). Additionally, there is little information available on the effects of anthropogenic land-use change on the ecology of the taxon. The works of Rowe-Rowe (1992) and Maddock and Perrin (1993) highlighted aspects of spatial use and habitat range of several mongoose species within KwaZulu-Natal Province (KZN), South Africa, in the late 1980s. There have been limited spatial ecological studies of the family since then.

**Table 3.1.** The home range sizes for 95% MCP estimate reported in previous literature for the three studied mongoose species (large grey, water and white-tailed) in Africa compared with the present current study.

| | | 95% MCP ($km^2$) | | |
|---|---|---|---|---|
| **Scientific name** | **Common name** | **Previous literature** | **Present study (± S.E)** | **References** |
| *Herpestes ichneumon* | Large grey mongoose | 1.35–2.77 | 7.8 ± 3.06 | Maddock, 1988; Rowe-Rowe, 1992; Skinner and Chimimba, 2005 |
| *Atilax paludinosus* | Water mongoose | 0.85–1.07 | 11.8 ± 1.52 | Maddock, 1988; Rowe-Rowe, 1992; Ray, J., 1997; Skinner and Chimimba, 2005 |

| | | | | |
|---|---|---|---|---|
| *Ichneumia albicauda* | White-tailed mongoose | 0.58–8.00 | 0.7 ± 0.10 | Taylor, 1970; Ikeda et al.,1982; Ikeda et al., 1983; Waser and Waser, 1985; Baker, 1997; Admasu et al., 2004 |

In a continually changing world where natural systems need to adapt, understanding the effects of anthropogenic land-use change on mesocarnivore dynamics and responses is of utmost importance. Therefore, understanding the tolerance of mongooses can act as a proxy for establishing models of how mesocarnivores react within a continually altering human-dominated landscape. Furthermore, gaining insight into the spatial ecology of mongooses can add valuable information to these deficient data taxa.

In our study, we determined the home range and core area utilisation of three co-existing mongoose species (large grey, water and white-tailed mongoose) within a natural and farmland mosaic in the KZN Midlands. We hypothesised that home range sizes of all three species of mongooses would be influenced by anthropogenic land-use change. The natural farmland mosaic of the KZN Midlands are ecologically highly productive; therefore, the resource needs of these three mongooses can likely be met in a relatively smaller range compared with previous literature. Alternatively, mongooses are restricted to smaller areas because of changing land-use. We predicted home range size of the respective species would be smaller in a natural and farmland mosaic than previously reported in protected areas (Rowe-Rowe 1992b; Admasu et al. 2004; Skinner and Chimimba 2005). The focus of this study was to fill knowledge gaps of the spatial ecology of the three mongoose species in an anthropogenically modified landscape. The research will contribute to the conservation requirements of these three mongoose species through improved ecological understanding.

## 3.3 Material and methods

*Study area*

Our study was undertaken at three different sites within the KZN Midlands: Dalcrue Farm (Nottingham Road) (29.381121° S, 29.907057° E), Mbona Private Nature Reserve (Karkloof) (29.290126° S, 30.360849° E) and Tillietudlem (Upper Dargle) (29.612649° S, 29.938602° E; Figure 3.1). Dalcrue is dominated by cattle grazing and cropland land-use type, with expansive natural grassland habitat. Mbona is surrounded by plantations and natural forest habitat. The land-use consists of 100 shareholder houses scatted throughout the nature reserve with habitat types of natural forest, grasslands and freshwater dams. Tillietudlem is dominated by grassland habitat with pockets of plantations, forest and bushlands. The study area is considered a conservation priority by Ezemvelo KZN Wildlife because of the rich diversity of flora and fauna species here despite continuing anthropogenic modification (Rowe-Rowe 1992b, 1994). The KZN Midlands experience the annual minimum and maximum temperatures of 9.2°C and 31.8°C, respectively (Ramesh and Downs 2015b). The area receives most of its 975.4 mm mean annual rainfall during the summer months (December–February), with frost occurring four months (May-August) of the year (Manson 1996). The topography of the area is characterised by a mix of undulating to rugged hill slopes with intermittent rivers and wetlands ≥ 1200 m a.s.l. Vegetation in the area is dominated by Highland Sourveld grassland, with clumps of indigenous forest (Killick 1990; Mucina and Rutherford 2006; Ramesh and Downs 2015b). The Karkloof area is dominated by intense levels of silviculture (*Pinus patula* and *Eucalyptus* spp.). Study sites were selected based on the presence of the target species and the willingness of landowners to provide access to their properties.

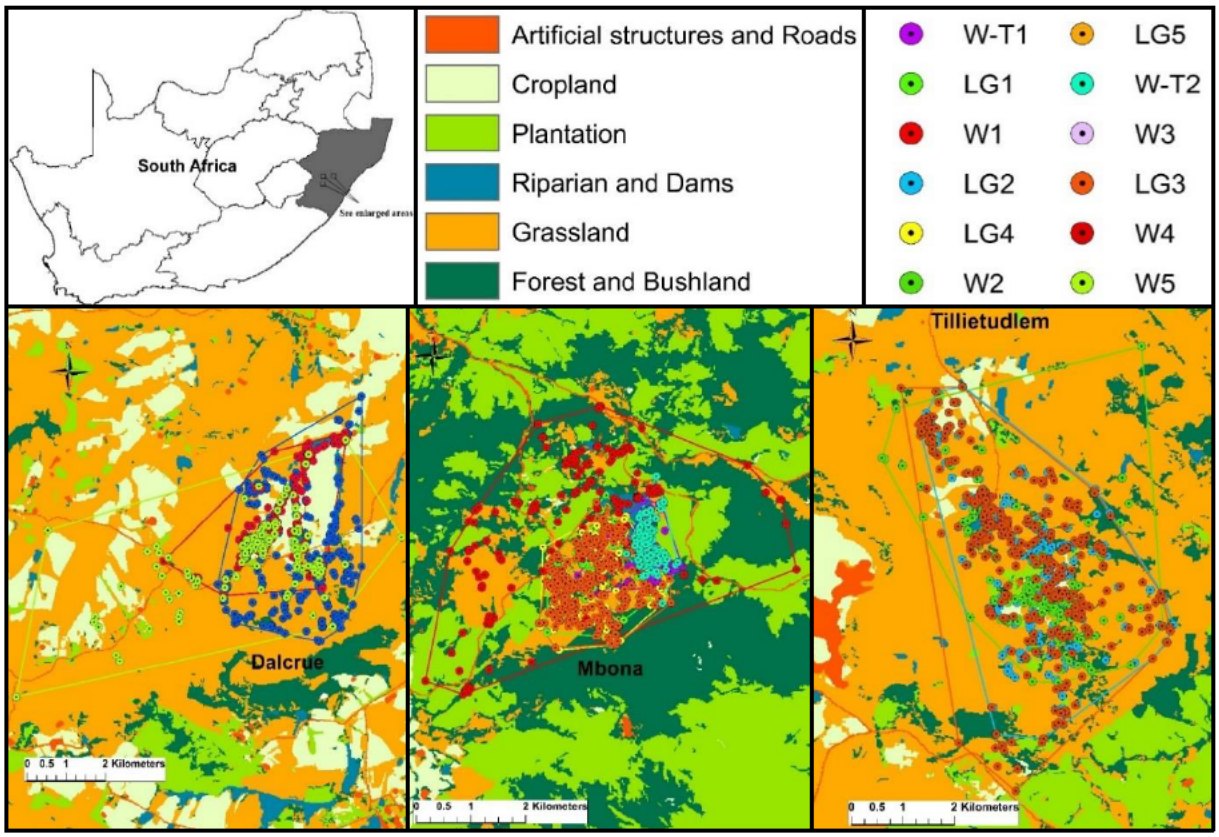


**Figure 3.1.** Location of the three study sites (Dalcrue, Mbona and Tillietudlem) in the KZN Midlands, KZN, South Africa; showing the reclassified main land-use types that characterise the area and distribution of GPS fixes for three mongoose species (large grey: LG, water: W and white-tailed: W-T).

Domestic livestock cattle (dairy and beef) (85%) and sheep (10%) dominate the domestic fauna of the KZN Midlands with horses (3%) and goats (2%) present to a lesser extent (Ramesh and Downs 2015b). Farmers privately own the bulk of the surrounding areas; however, fragmented portions of the landscape are owned by the government and several external organisations. Most privately-owned farmland boundaries are fenced internally and externally with cattle wire fences, essential for the implementation of rotational grazing camps. Fencing of this nature allows the free passage of small wildlife between adjacent areas. The study area is inhabited by several diverse naturally occurring wildlife species including small

and medium-sized carnivore, ungulate, rodent and bird species (Ramesh and Downs 2015b). Large carnivores (> 15 kg) were extirpated from the region before 1900 (Skinner and Chimimba 2005).

*Data collection*

We captured and collared individuals of the three mongoose species (large grey, water and white-tailed) between August 2016 and May 2017. Trap and camera trap locations were placed based on areas frequented by the target species, which were lured with bait. Step plate traps had dimensions 50 x 50 x 100 cm. Five traps were rotationally set in dense vegetative areas close to water sources and grasslands (wetlands, vleis and streams) and baited with a combination of chicken hearts and chicken mala. The traps were camouflaged with surrounding indigenous vegetation (grasses, branches and surrounding trees) because of a high failure rate associated with uncovered traps (Maddock 1988). We set traps each morning and checked them in the afternoon, and again the following morning because of differential activity period of the respective target species. Trap success was calculated using the number of individuals captured divided by the total effort, multiplied by 100 (Caceres et al. 2011).

A veterinarian sedated each trapped mongoose with a mixture of Anaket (0.8 mg/kg, Bayer, South Africa) and Domitor (0.5 mg/kg, Pfizer, South Africa) based on the individual's approximate body mass. Once completely immobilised, individuals were removed from the trap, sexed and morphometric measurements were taken. These included: total length, body length, chest girth, foot length, canine length, body mass and head length. Individuals which met the minimum weight requirement (collar mass < 3% of body mass) (Boitani and Fuller 2000; Kenward 2001) were fitted with a collar with a GPS-GSM/UHF transmitter (Animal Trackem, Pietermaritzburg, South Africa) weighing ~52 g. A finger gap was left under each collar to prevent discomfort and allow for additional neck growth while at the same time

preventing removal or excessive movement of the device. Once the individual mongoose was collared, it was given a reversal anaesthetic of Antisedan (0.5 mg/kg, Zoetis, South Africa). Four trapped individuals (one large grey mongoose and three water mongooses) had to be released without a collar because they did not meet the minimum weight requirements. Ethical clearance to trap, sedate and GPS/UHF collar individuals of the three species of mongooses was provided by the University of KwaZulu-Natal (No. 020/15/animal). A portable solar-powered based station receiver was permanently set up at each study site to download telemetry data from the GPS-GSM/UHF transmitter. Data from the base station receiver were continually sent via the cellular network (GSM) and later accessed via Wireless Wildlife as a *csv. document.

*Data screening*

Home range estimations are affected by the number and precision of GPS fixes. That is, small terrestrial animal behaviour is associated with higher inaccuracy and failed GPS fixes when compared with larger-bodied animals (Laver et al. 2015). Errors in GPS fixes are pronounced in small animals through reduced satellite reception, which generally diminishes GPS fix accuracy (Laver et al. 2015). Before any home range analyses, data were screened using packages adehabitatLT' version 0.3.20, 'adehabitatMA' version 0.3.10, 'ade4' version 1.7e4 and 'sp' version 1.2e3 in R software (version 3.1.2) to remove inaccurate data points (Calenge 2006; RStudio 2015; Drabik-Hamshare and Downs 2017). Autocorrelation generally results from a lack of statistical independence between subsequent GPS points in both time and space (Legendre 1993). This violation typically results from too frequently obtained GPS fixes over a short time interval. To reduce the effects of autocorrelation of GPS points, GPS fixes were scheduled to record a geographic location point at 4-h intervals during 12-h periods for all tagged mongoose individuals. As a consequence of activity time differences between the

three species (large grey mongoose diurnal; water and white-tailed mongoose species nocturnal) GPS fixes were set to record from 06:00–18:00 for the large grey mongoose and 18:00–06:00 for water and white-tailed mongoose species.

*Home range and core area utilisation analyses*

We imported the GPS coordinated data into ArcGIS 10.4 (ESRI, Redlands, CA, USA), and they were projected in UTM (WGS 1984 UTM Zone 35S and 36S). We determined home range estimations following the criteria set by Laver and Kelly (2008). We estimated home range size using three home range methods: Kernel Density Estimate method (KDE), Maximum Convex Polygon (MCP) and Local Convex Hull (LoCoH). R package rhr was used in user interface R studio (1.2.909 ) to estimate 50% (core area utilisation estimates) and 95% from the three home range estimate methods (RStudio 2015; Signer and Balkenhol 2015). To date, there is no optimal smoothing parameter for KDE, thus, we used the reference bandwidth smoothing parameter (href) to prevent over-smoothing and excessive fragmentation of home ranges (Walter et al. 2011). The LoCoH method constructs a convex hull around each point and the points of its nearest neighbour (*n*) (Getz et al. 2007). UD surfaces are affected by the *h* and *n* parameters, notability high values of *h* or *n* generate uniform UD surfaces, reducing variability, whereas smaller values increase the resolution of valleys and peaks through a more precise fit of the dataset (Worton 1989; Fieberg 2007; Lichti and Swihart 2011). Buffer and resolution levels were manually manipulated based on visual assessment (Drabik-Hamshare and Downs 2017). We performed repeated-measures analyses of variance (RMANOVA) for large grey mongoose, water mongoose and white-tailed mongoose using Statistica (Statsoft Inc., Tulsa, OK, USA) for the three estimation methods to delineate differences in the measured home range sizes at both the 50% and 95% levels using the three home range methods. We established site fidelity for each individual using the Mean Square Distance (MSD) and Linearity index (LI)

from Centre of Activity using the 'rhr' version 1.2.909 package in R (Signer and Balkenhol 2015). Site fidelity was tested using 10,000 bootstrap replicates at 95% confidence interval.

*Home range overlap*

Spatial overlaps of mongoose home ranges were estimated through static analyses outlined in Kernohan et al. (2001). Home range measures of MCP and KDE were used to estimate home range overlap at both the core area 50% and 95% level. We calculated spatial overlap for all collared mongooses in the three study sites. Spatial overlaps were only estimated for mongooses that were tracked over a parallel time period (exception individual LG1 whose device failed after 47 days). Spatial overlaps of the mongooses' home ranges represented an estimate because only a proportion of the population of mongooses was collared and tracked between the sites (Carter et al. 2012)

**3.4 Results**

Trap success rate varied between the three sites and ranged between 1.2–5.2 %. Twelve mongooses were collared (five large grey mongooses, five water mongooses and two white-tailed mongooses). Tagged large grey consisted of four females and one male, water mongooses comprised of three females and two males and white-tailed mongoose only females. A total of 4142 GPS locations were cumulatively gathered between all the tagged mongooses. The number of GPS locations per individual ranged from 102–591 after filtering, during a tracking period that spanned 47–212 days (Table 3.2). A hardware issue caused the LG1 tag to cease functioning after only 47 days.

**Table 3.2** The core area uses of three mongoose species (large grey: LG, water: W and white-tailed: W-T) collared with GPS/UHF transmitters in the fragmented natural habitat and farmland mosaic of the KZN Midlands, South Africa. The core area 50% confidence estimates for MCP, KDE and LoCoH (KDE and LoCoH methods utilised smoothing parameters reference bandwidth (href) and default k turning parameter, respectively).

| Individual | Species | Sex | Age class | Start Date | End Date | No. of GPS fixes | No. of Days | Mass (kg) | Core area ($km^2$) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | MCP | KDE (href) | LoCoH (k) |
| LG1 | Large grey | F | Adult | 7/11/2016 | 25/12/2016 | 102 | 47 | 3.75 | 0.4 | 0.7 (197.3) | 0.1 (10) |
| LG2 | Large grey | F | Adult | 8/12/2016 | 24/6/2017 | 591 | 199 | 3.57 | 3.0 | 3.4 (370.0) | 0.5 (24) |
| LG3 | Large grey | F | Sub-adult | 24/04/2017 | 17/08/2017 | 298 | 113 | 2.03 | 1.0 | 1.1 (300.1) | 0.4 (17) |
| LG4 | Large grey | F | Sub-adult | 26/04/2017 | 17/08/2017 | 299 | 113 | 2.2 | 0.8 | 1.0 (183.5) | 0.3 (17) |
| LG5 | Large grey | M | Sub-adult | 15/03/2017 | 16/09/2017 | 475 | 186 | 2.09 | 3.3 | 4.6 (445.0) | 0.7 (22) |
| W1 | Water | M | Adult | 11/11/2016 | 21/3/2017 | 248 | 131 | 3.52 | 4.5 | 5.5 (492.4) | 0.4 (16) |
| W2 | Water | F | Adult | 23/01/2017 | 16/7/2017 | 457 | 174 | 2.94 | 0.8 | 1.1 (312.6) | 0.1 (21) |
| W3 | Water | M | Adult | 13/03/2017 | 10/10/2017 | 445 | 212 | 3.5 | 4.2 | 5.3 (405.3) | 0.7 (21) |
| W4 | Water | F | Sub-adult | 30/03/2017 | 06/102017 | 413 | 190 | 2.76 | 0.8 | 1.2 (285.9) | 0.04 (20) |
| W5 | Water | F | Adult | 4/4/2017 | 18/8/2017 | 212 | 136 | 3.02 | 0.9 | 2.3 (409.5) | 0.2 (15) |
| W-T1 | White-tailed | F | Adult | 30/09/2016 | 26/4/2017 | 420 | 209 | 4.5 | 0.2 | 0.2 (89.5) | 0.02 (20) |
| W-T2 | White-tailed | F | Sub-adult | 13/03/2017 | 30/7/2017 | 182 | 140 | 4 | 0.1 | 0.2 (98.2) | 0.02 (13) |

*Home range and core area utilisation*

We accumulated a total of 1775, 1756 and 602 GPS fixes for large grey, water and white-tailed mongoose over the study period. Only LG2 exhibited a lack of site fidelity; this may be because of an arson event that took place on the Tillietudlem study site (B. Roth pers. comm. 2017), causing the individual to alter its natural behaviour in search of resources and suitable habitat.

The mean (± S.E.) 95% confidence home range sizes for mongooses varied between the three species. Large grey mongooses had intermediate home range sizes (MCP 7.8 ± 3.06 $km^2$, KDE = 9.9 ± 3.77 $km^2$, LoCoH = 4.5 ± 1.75 $km^2$, Figure 3.2b). Water mongoose had the largest mean home range across the species (MCP = 11.8 ± 1.52 $km^2$, KDE = 15.6 ± 2.70 $km^2$, LoCoH = 4.6 ± 0.78 $km^2$, Figure 3.2c). White-tailed mongooses had the smallest mean home range size regardless of home range estimation method (MCP = 0.7 ± 0.10 $km^2$, KDE = 0.9 ± 0.06 $km^2$, LoCoH = 0.4 ± 0.01 $km^2$, Figure 3.2a).

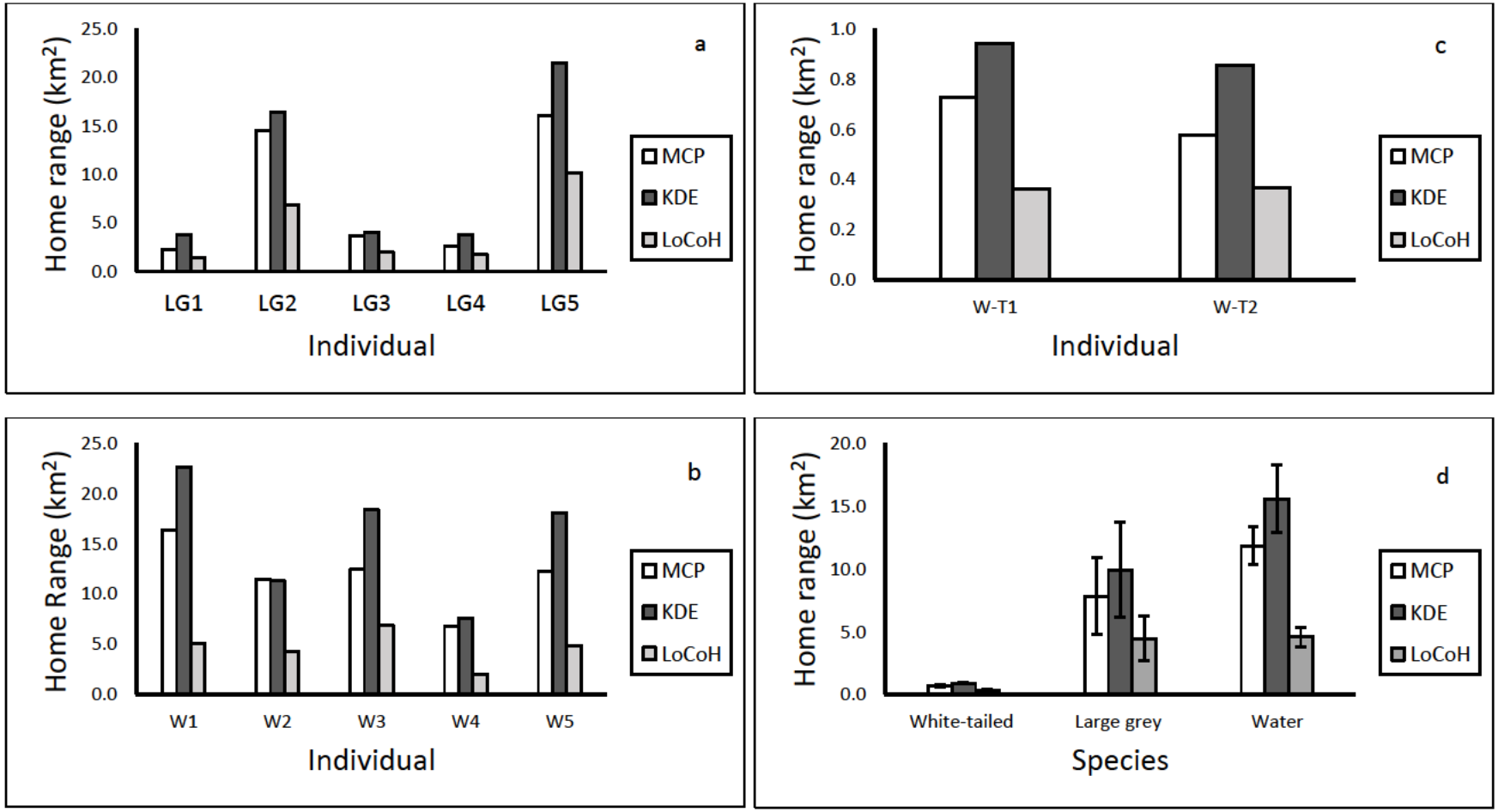


**Figure 3.2.** a. Individual home ranges of the three mongoose species (a. large grey mongoose (LG, n = 5), b. water mongoose (W, n = 5) and c. white-tailed mongoose (W-T, n = 2)); and d. overall mean home range (± S.E.) size for the respective species using the 95% confidence interval of the three home range estimates (MCP, KDE, LoCoH) in the present study.

Home range sizes differed significantly according to the estimator used between large grey mongoose, water mongoose and white-tailed mongoose for the 95% home range estimates (RMANOVA, Current effect: $F_{(4,18)} = 4.168$, $P = 0.015$). A posthoc Scheffe test showed significant differences between the home range estimation methods and species main effects for the home range estimation between MCP and LoCoH ($P = 0.025$) and KDE and LoCoH ($P = 0.0003$). This was only true for the water mongoose. The small sample size within species and the differential period between collaring individuals precluded accurate statistical comparisons between sexes and seasons within and between species.

Core area estimates for MCP, KDE and LoCoH estimations varied between the three mongoose species and between individuals within each species (Supplementary information Figures S3.1-3; Figure 3.2). The mean (± S.E.) 50% confidence core area utilisation range sizes for mongooses differed between the three species. Large grey mongooses had intermediate core area utilisation range sizes (MCP 1.7 ± 0.60 $km^2$, KDE = 2.2 ± 0.77 $km^2$, LoCoH = 0.4 ± 0.10 $km^2$). Water mongooses had the largest mean home range across the species except for the LoCoH method (MCP = 2.2 ± 0.86 $km^2$, KDE = 3.1 ± 0.96 $km^2$, LoCoH = 0.3 ± 0.12 $km^2$). White-tailed mongooses had the smallest mean core area utilisation range sizes regardless of home range estimation method (MCP = 0.1 ± 0.01 $km^2$, KDE = 0.1 ± 0.01 $km^2$, LoCoH = 0.2 ± 0.02 $km^2$. Core home range sizes did not differ significantly according to the estimator used between large grey mongoose, water mongoose and white-tailed mongoose for the 50% home range estimates (RMANOVA, Current effect: $F_{(4,18)} = 1.683$, $P = 0.198$, Supplementary information Figures S3.1-3)

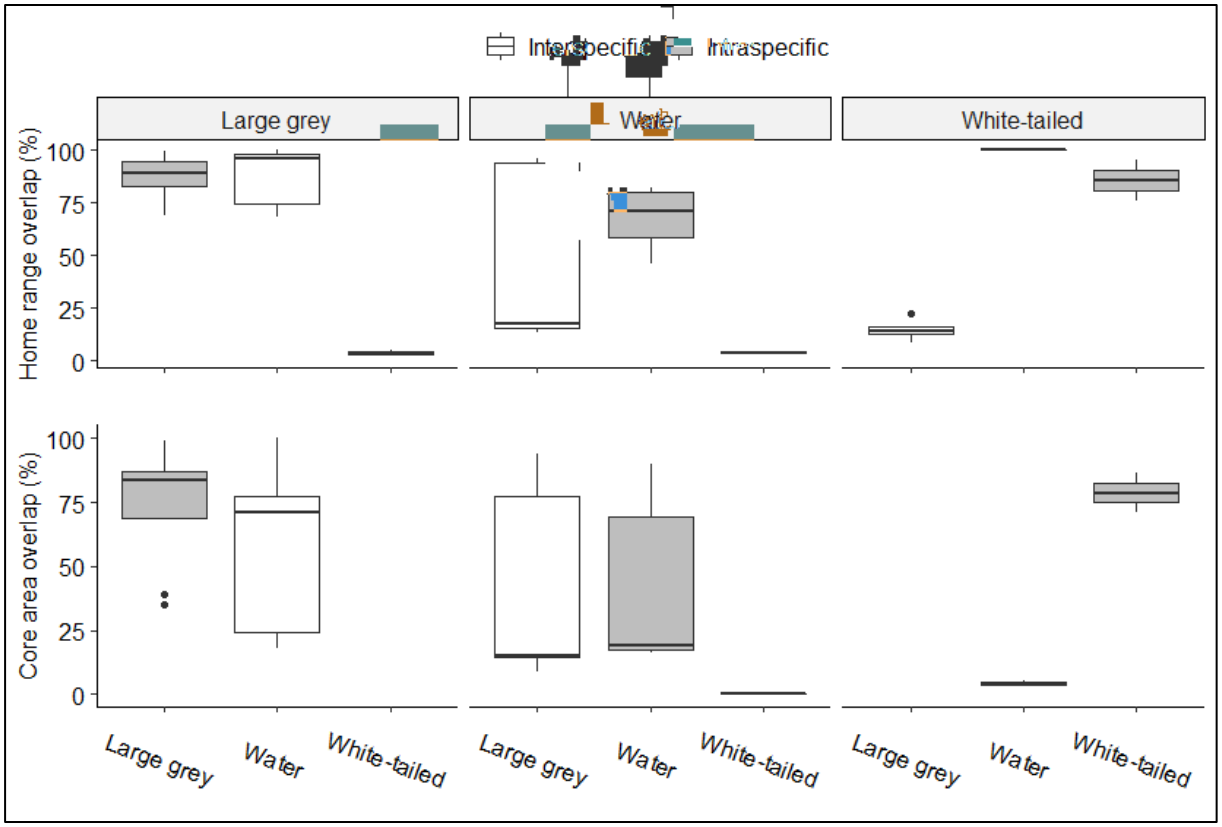


**Figure 3.3.** The interspecific and intraspecific home range and core area overlap between three mongoose species (large grey: LG, water: W and white-tailed: W-T) using MCP home range estimates of collared mongooses in the present study.

Degree of intraspecific home range overlap between collared mongooses varied between the species (95% mean MCP: large grey 88 ± 10.6%, water 71 ± 19.2%, white-tailed 85 ± 13.6; Figure 3.3). Degree of intraspecific core area overlap of collared mongooses was lower than home range overlap (50% mean MCP: large grey 74 ± 23.8%, water 41 ± 36.1%, white-tailed 78 ± 11.1%; Figure 3.3).

Degree of interspecific home range and core area overlap for the 95% and 50% MCP values differed between the collared mongooses. Large grey mongooses had high to intermediate levels of overlap with water mongooses for home range (HR) and core area (CA) (HR: 87 ± 15%, CA: 58 ± 35.5%), but had low levels to no overlap with white-tailed mongooses

(HR: 4 ± 1.3%, CA: 0%). Water mongooses had an intermediate level of overlap with high variation with large grey mongooses for both HR and CA measures (HR: 47 ± 43.5%, CA: 42 ± 40.4%), yet had low levels of overlap with white-tailed mongooses for both HR and CA measures (HR: 4 ± 0.6%, CA 0.1 ± 0.03%). White-tailed mongooses had low levels to no overlap with large grey mongooses (HR: 14 ± 4.6%, CA 0%), but had high levels of overlap for HR water mongoose but no overlap was detected present for CA (HR: 100%, CA 0%).

Home range shapes using the MCP and KDE estimates varied between the three species of mongooses. Large grey mongooses and water mongooses had large, variable shaped home ranges that encompassed large portions of land-use features (grasslands and riparian and dam). White-tailed mongooses had reduced home range shapes, which were generally close to anthropogenic artificial structures and road dominated land-use type.

### 3.5 Discussion

The present study highlighted the intraspecific and interspecific differences observed in the home range size and core area utilisation of the three mongoose species in the fragmented landscape of the KZN Midlands, South Africa which is mainly under agricultural land-use. Emerging trends in the data provide new information for each of these three mongoose species on the home range and core area utilisation, although analyses between sexes, age and seasons could not be established. The present study found inter-individual differences which are likely a result of life stage, physiological condition (sex, size, energetic state, age) and behavioural state (competition and foraging strategies) of an individual (Clutton-Brock and Harvey 1978; Maddock 1988; Nathan et al. 2008; Cagnacci et al. 2010). The individual state was typically unique for each tracked animal. Therefore, additional factors (landscape type, food availability, seasonality, level of fragmentation) must be considered.

Contrary to our predictions, home range sizes of large grey mongoose and water mongooses in the present study were greater than home range estimates in natural protected areas in previous studies (Table 3.1) (Maddock 1988; Rowe-Rowe 1992b; Skinner and Chimimba 2005). In Vernon Crookes Nature Reserve, KZN, South Africa, Maddock (1988) estimated the 95% MCP home ranges as 1.35–2.77 $km^2$ for the large grey mongoose (n = 4) and 0.85–1.07 $km^2$ for the water mongoose (n = 2). These home range size differences highlight that both species may have lower resource availability in the anthropogenically fragmented environment compared with natural protected areas (Altieri 1999; Viswanathan et al. 2011). Although the biotic and abiotic conditions between Vernon Crookes Nature Reserve and the KZN Midlands differ, the findings of the present study illustrate two possible explanations: either previous knowledge underestimated the home range requirements for these mongooses, or resources for these two species of mongooses are more limited in a natural and farmland mosaic. When resources are scarce and patchily distributed, animals will have to traverse an extensive range to obtain resources to ensure individual fitness (Clutton-Brock and Harvey 1978; Boutin 1990).

Additionally, the factor of improved telemetry technology has allowed for more accurate and continuous data collection of tracked individuals (Hebblewhite and Haydon 2010; Tomkiewicz et al. 2010). Interestingly the white-tailed mongoose home range sizes were similar to white-tailed home range measures in other studies (Table 3.1**)**. The small home range sizes suggest that resource richness is likely sufficient for white-tailed mongoose in the study site or that their diet breadth is wider (Admasu et al. 2004; Skinner and Chimimba 2005).

Urbanisation is arguably one of the most extreme forms of anthropogenically altered landscapes. Numerous carnivore species have consequently modified their home range sizes, territories and habitat preferences in urban areas or faced population collapses (Glen and Dickman 2005; Ritchie and Johnson 2009; Bateman and Fleming 2012). In accordance with

the above, Admusa (2004) suggested the white-tailed mongoose has the necessary behavioural plasticity to adapt to human-modified landscapes. This mongoose species has been noted to scavenge through human refuse and benefit from artificial lighting to hunt nocturnal invertebrates (Admasu et al. 2004). The white-tailed mongoose has the potential to become a peri-urban or urban exploiter. This adaptive behaviour characteristic may potentially lead to increased human–wildlife conflict in residential areas (that have chicken runs, exposed refuse) as landscapes become increasingly fragmented, and resources diminish. Overall, home range sizes were larger in the fragmented natural habitat and farmland mosaic of the KZN midlands compared with natural systems where the ecology of both large grey mongoose and water mongoose have been studied. However, white-tailed mongooses home range sizes for the present study aligned with previous literature occurring in natural systems (Table 3.1).

Relationships between site resources and habitat availability have been shown to be determinants of home range size variability within mammals (Boutin 1990; Šálek et al. 2015). The evident intraspecific differences between large grey mongoose and water mongoose individuals contribute to the expanding body of literature for interindividual variation that exists (White and Walker 1997; Fryxell et al. 2014). The social status of dependent juveniles, unique personalities and thus unique behavioural responses to environmental conditions have been used to explain variation in home range within a species. As such, without an extensive study that addresses all the above factors, it would be unwise to offer definitive reasons for the variation detected in home range sizes.

The core area utilisation by the mongooses varied in size and shape for all three estimates for the tagged individuals. Water mongoose individuals showed the largest core area for the KDE method, followed by large grey and white-tailed mongooses. However, for the LoCoH method, large grey mongooses showed the largest core area range, followed by the water and white-tailed mongooses. Identification of these intensely utilised areas indicated

areas of high dependence and activity for each species. The core area used by each collared individual was generally disproportional, indicating favouring areas of higher resource availability. The disproportional use within home ranges highlights the heterogeneous nature of resource distribution across space and time. This may be exacerbated in anthropogenically modified landscapes. Individuals' home ranges should include areas that maximise individual fitness whilst limiting incurred costs (Powell 2000). The differential niche-use and activity of the large grey, water and white-tailed mongooses allow the species to co-exist in these anthropogenically modified and fragmented landscapes. Therefore, the three species can co-exist in areas consisting of suitable habitats that are abundant in resources, with relatively low predation risk.

The three collared mongoose species shared a high degree of overlap of their core area 50% and 95% home range with both collared conspecifics and interspecific mongooses. There was a lack of territoriality between female conspecifics collared mongooses, which showed that adult females and sub-adult females were typically not territorial. Females of the present study's mongoose species are known to form small species-specific groups when sub-adults fail to disperse from their natal range (natal philopatry) (Waser and Waser 1985; Skinner and Chimimba 2005). It is suggested that white-tailed mongoose can be gregarious when food resources are abundant as it reduces intraspecific competition whilst promoting safety from predators (Macdonald 1983; Waser and Waser 1985; Admasu et al. 2004). No adult, male large grey mongoose, water mongoose or white-tailed mongoose were collared and simultaneously tracked in the same site during the study (Dalcrue, Mbona or Tillietudlem). Adult male mongooses of these three species are known to be highly territorial and their core home range areas overlap with numerous female ranges, but rarely with males (Palomares and Delibes 1993; Admasu et al. 2004; Skinner and Chimimba 2005). The high spatial overlap between the three species of collared mongooses at each site in the present study emphasised the presence of niche

partitioning among female mongooses (Maddock 1988). The ecological difference (temporal and dietary) between the species of mongooses allows them to coexist and generally avoid competition (Maddock 1988). Therefore, niche partitioning in these two dimensions enables these species of mongooses to have a high degree of home range overlap.

The present study highlighted the behavioural plasticity and adaptability present in these three co-existing mongoose species. A highly advanced system of co-use seems to be facilitating inhabitation, of three mongoose species in this anthropogenically modified and fragmented environment. In addition to this, the three species of mongooses illustrated individual differences in how they navigate the landscape. Home range sizes for large grey and water mongoose of this study emphasised the importance of telemetry advancements and studies in anthropogenically modified landscapes in accurately reflecting the spatial ecology of tracked animals.

To achieve effective conservation management of these three species of mongooses requires greater understanding in spatial ecology (movement, habitat use and home range size). Our study was the first attempt since the 1980s to investigate the spatial ecology of these three mongooses in KZN and the first to establish the effects of anthropogenic land-use change on the spatial ecology of these three large species of mongoose in KZN and southern Africa. The current study has therefore contributed to understanding the home range and core area utilisation of the study species within a continually anthropogenically modified and changing environment. We recommend that further research investigates the effects of urbanisation on the species by examining mongooses spatial ecology in a natural habitat vs. urban mosaic landscape, and so contribute more to understanding their persistence over an anthropogenic land-use gradient. Furthermore, future studies should investigate how these species can niche partition within highly fragmented landscapes, allowing them to coexist and avoid competition (Streicher et al. in prep.). In conclusion, these three species of mongooses are highly adaptive,

and this enables them to exist in the anthropogenically modified landscape of the present study. However, there still exists an evident data deficiency on the Herpestidae family which requires further attention.

### 3.6 Acknowledgements

We would like to thank the National Research Foundation (ZA) (SFH180530337163; 98404), the Oppenheimer Memorial Trust (20616/01) (ZA), the Hans Hoheisen Charitable Trust (ZA) and the University of KwaZulu-Natal (GC Weightman Scholarship) (ZA) for their financial support. The Ford Wildlife Foundation (ZA) kindly provided vehicle support. Veterinarians Dr M. De Villiers, Dr D. Gibbs, Dr R. van Deventer are thanked for their continued assistance and advice during immobilising and collaring of the three mongoose species. We would also like to acknowledge the land managers: B. Roth of Tillietudlem, G. Powell and D. Forsyth of Mbona Nature Reserve and J. von Rooyen of Dalcrue; who allowed us to work and capture mongooses on their properties. Finally, JPS would like to thank M. Streicher for her assistance with fieldwork and input during the writeup of the study. We are grateful for the constructive comments of the reviewers.

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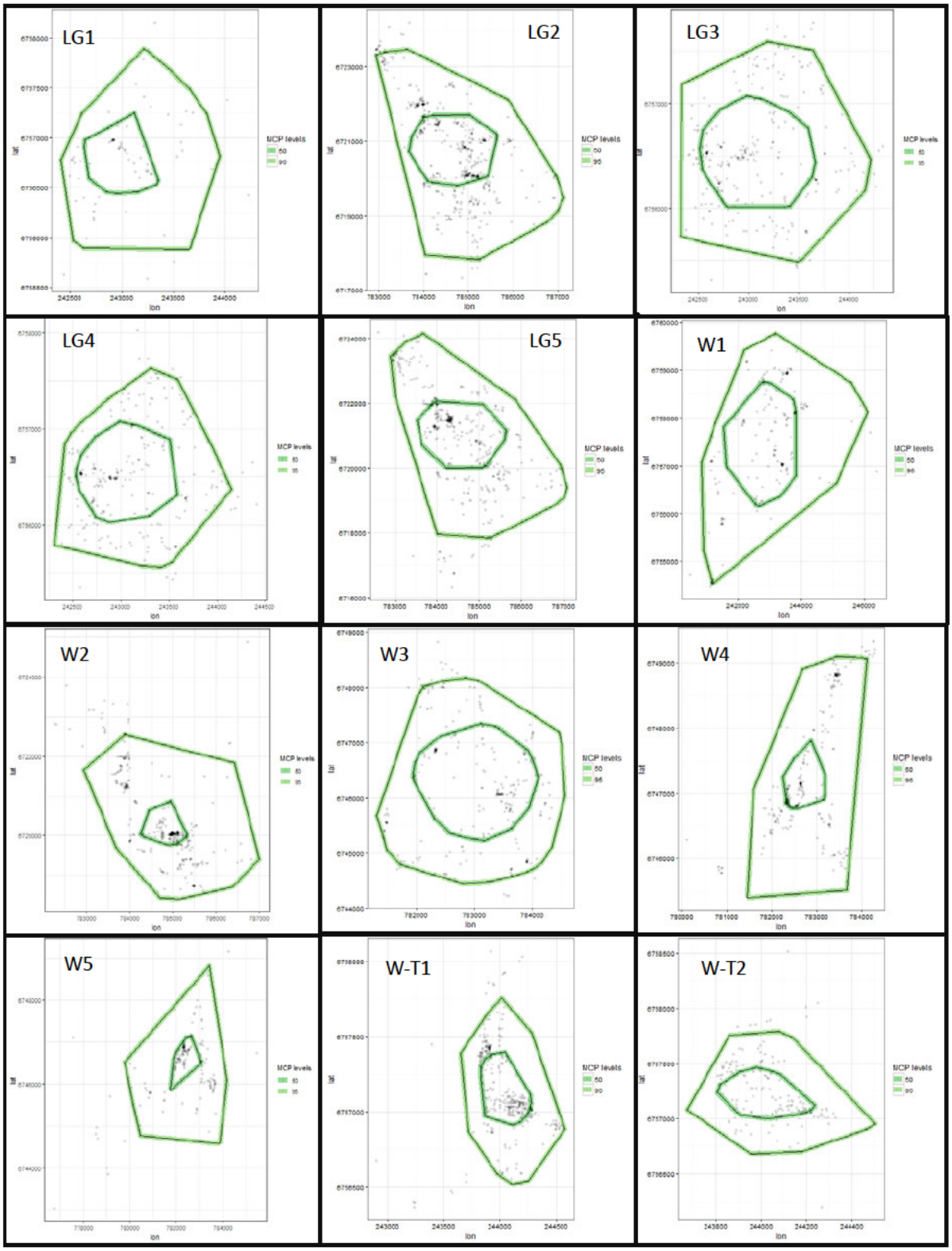
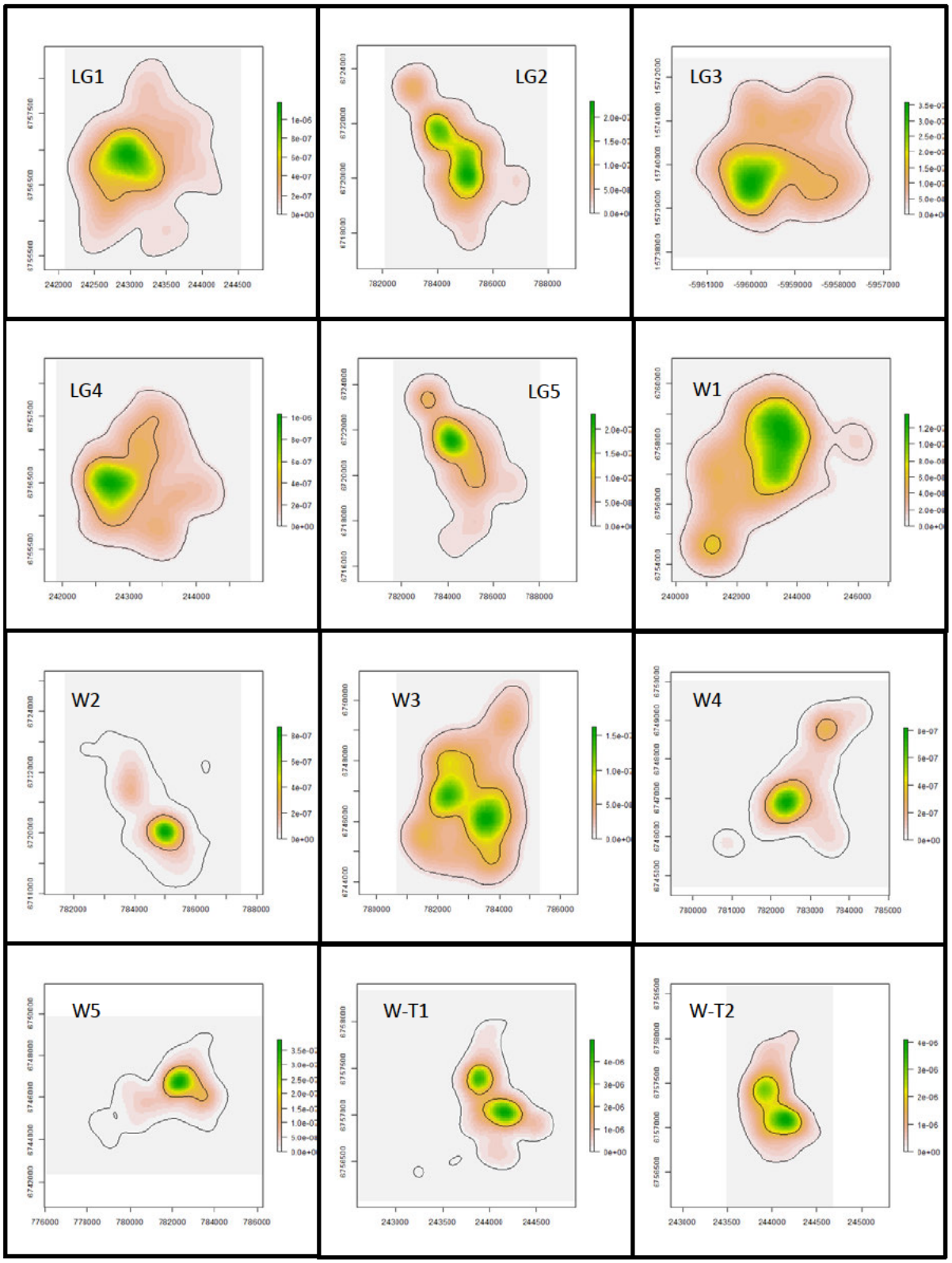
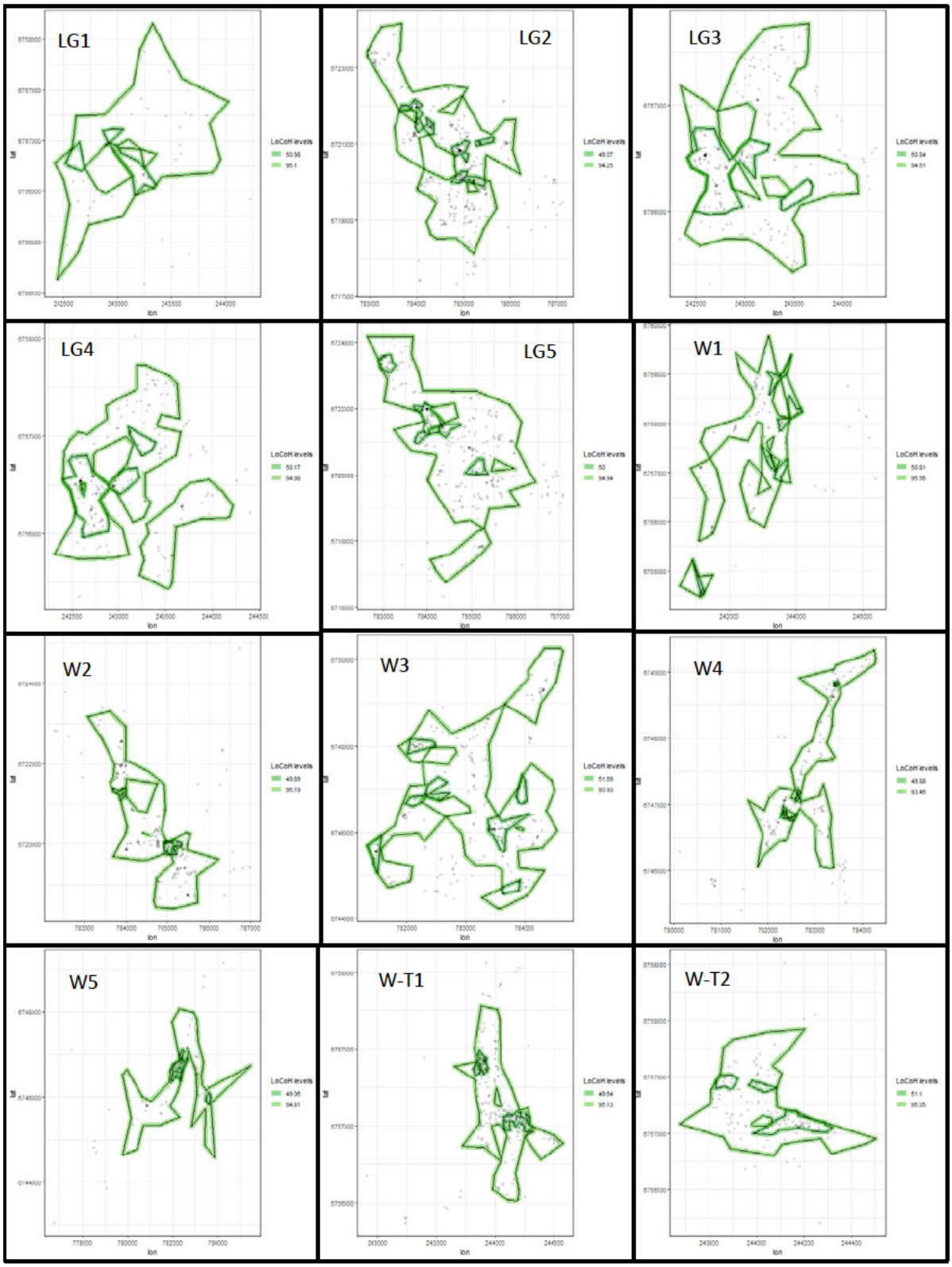
### 3.8 Supplementary information




**Supplementary information Figure S3.1.** The 50% core and 95% MCP estimations for three mongoose species (large grey: LG, water: W and white-tailed: W-T) collared with GPS/UHF transmitters in the present study.




**Supplementary information Figure S3.2.** The 50% core and 95% KDE estimations for three mongoose species (large grey: LG, water: W and white-tailed: W-T) collared with GPS/UHF transmitters in the present study.




**Supplementary information Figure S3.3.**The 50% core and 95% LoCoH estimations for three mongoose species (large grey: LG, water: W and white-tailed: W-T) collared with GPS/UHF transmitters in the present study.

**Supplementary information Table S3.1.** The percentage of home range area overlap of three mongoose species (large grey: LG, water: W and white-tailed: W-T) using two home range estimates MCP and KDE at the 50% and 95% confidence interval that mongooses shared with other radio-collared mongooses and radio-collared conspecifics in the present study. (number of mongooses sharing the home range).

| | | Percentage of home range overlap | | | | |
|---|---|---|---|---|---|---|
| **Individual** | **Site** | **MCP 95% Conspecifics** | **MCP 50%** | **MCP 95%** | **KDE 50%** | **KDE 95%** |
| LG1 | Mbona | 93 (2) | 100 (3) | 100 (5) | 99 (3) | 100 (5) |
| LG2 | Tillietudlem | 99 (1) | 93 (2) | 100 (2) | 97 (2) | 99 (2) |
| LG3 | Mbona | 87 (2) | 91 (3) | 100 (5) | 94 (3) | 99 (5) |
| LG4 | Mbona | 100 (2) | 100 (3) | 98 (5) | 100 (3) | 100 (5) |
| LG5 | Tillietudlem | 90 (1) | 79 (2) | 100 (2) | 65 (2) | 78 (2) |
| W1 | Mbona | N/A | 17 (5) | 16 (5) | 18 (5) | 35 (5) |
| W2 | Tillietudlem | N/A | 94 (2) | 96 (2) | 94 (2) | 100 (2) |
| W3 | Dalcrue | 79 (2) | 36 (2) | 100 (2) | 39 (2) | 69 (2) |
| W4 | Dalcrue | 99 (2) | 90 (2) | 99 (2) | 82 (2) | 98 (2) |
| W5 | Dalcrue | 71 (2) | 86 (2) | 71 (2) | 89 (2) | 65 (2) |
| W-T1 | Mbona | 76 (1) | 71 (2) | 100 (5) | 91 (2) | 100 (5) |
| W-T2 | Mbona | 94 (1) | 89 (2) | 100 (5) | 94 (2) | 100 (5) |

# CHAPTER 4

## An African urban mesocarnivore: Navigating the urban matrix of Durban, South Africa

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**Running header***:* A mongoose species in an urban matrix

## 4.1 Abstract

Urbanisation is rapidly transforming natural habitats with the potential to benefit synanthropic species, especially mesocarnivore species. Knowledge of the spatio-temporal ecology of mongoose species in an urban matrix is limited. Consequently, we examined the home range and habitat use of water mongoose (*Atilax paludinosus*, n = 14) in an urban matrix landscape. Mongooses were collared and radio-tracked using Global Positioning System (GPS)-Ultra high frequency (UHF) transmitters between July 2018 and October 2019 in the Upper Highway area's urban matrix, eThekwini Municipality, Durban, KwaZulu-Natal, South Africa. Habitat selection for water mongoose indicated that the species avoided built-up urban areas favouring fragments of forest and bushland habitat types close to natural water sources within the urban mosaic. The adaptability and generalist nature of water mongoose has facilitated their persistence in the urban mosaic. Variability in the home range size and habitat use of individuals of the water mongooses highlights their ability to adapt to anthropogenically modified landscape, yet they were highly dependent on natural refugia in this urban matrix.

***Keywords:*** Herpestidae; Urban land-use; GPS telemetry; Small carnivore; Home range; Habitat use; South Africa

## 4.2 Introduction

Urbanisation has increased at an unprecedented rate globally and is predicted to continue to accelerate in developing countries (United Nations 2019). At the current rate of urban expansion, it is projected that 70% of the human population will live in an urban setting by 2050 (Seto and Shepherd 2009; United Nations 2019). Gert (2010) defined an urban area as "an area of human residence, activity, and the associated land area developed for those purposes". Usually characterised by a threshold human density, urbanisation exerts profound pressures on the pre-existing natural landscapes. The process of urbanisation results in extensive habitat fragmentation, isolation and degradation (Haddad et al., 2015; Marzluff 2001). It alters nutrient cycling and energy flow (Grimm et al., 2008; McDonnell et al., 1997), and hydrology (Booth and Jackson 1997; McGrane 2016) through the homogenising transformation of flora and faunal communities by the anthropogenic conversion for commercial, industrial or residential use (Amstrup et al., 2004; Blair 2004; McKinney 2006; Seto et al., 2011). Urbanisation has critical implications for native biodiversity and ecosystem function, which impacts local and global biodiversity (Allan et al., 2015; McKinney 2002; Pickett et al., 2001; Seto et al., 2011).

Animal diversity in urban areas is often homogenised (McKinney 2008). However, faunal responses to urbanisation vary between species (McPherson et al., 2016; Pickett et al., 2001; Ponsonby and Schwaibold 2019; Thabethe and Downs 2018; Widdows and Downs 2015; Zungu et al., 2020). Site-specific characteristics of urban design play a role in determining the faunal response within the urban matrix (Alberti and Marzluff 2004; Gehrt et al., 2010; Pickett et al., 2001; Yirga et al., 2016). Most species avoid urban areas, termed "urbanophobes", because of the adverse effects on individual fitness through anthropogenic stress, competition for resources or diminution of resources (Bateman and Fleming 2012; McKinney 2002; Meillere et al., 2015; Wittig et al., 1985). However, several species are noted to persist ("urban

adapters"), and some cases dominate ("urban exploiters") the anthropogenic urban mosaic landscape (McKinney 2002). A generalist nature is typically characterised. These "synanthropic" species behavioural and demographic plasticity, which predisposes them to inhabit the human landscape at the suburban and urban level (Bateman and Fleming 2012; Lowry et al., 2013; Widdows and Downs 2016). Synanthropic species typically exhibit "urban" behavioural changes such as increased population densities, altered diel movement patterns and reduced dispersal and home range size when compared with conspecifics in natural habitats (Lowry et al., 2013; Šálek et al., 2015).

Several mammalian mesocarnivores are recorded as urban adapters or "synanthropic species" (Gehrt et al., 2010; Lesmeister et al., 2015; Šálek et al., 2015). Mesocarnivore species thriving in urban mosaic landscapes are suggested to be tolerant of human activities and typically show behavioural plasticity (e.g. habitat use, home range, diet and movement) (Bateman and Fleming 2012; Ponsonby and Schwaibold 2019; Šálek et al., 2015; Widdows and Downs 2015; Yirga et al., 2016). Synanthropic species must additionally be tolerated and perceived as non-threatening to humans to avoid additional persecution pressures (Gehrt et al., 2009; Gehrt et al., 2010). These traits are crucial for carnivores to colonise, persist or thrive in urban ecosystems (Bateman and Fleming 2012). Within urban mosaic landscapes, mesocarnivore species can generally exploit the urban attributes of increased food availability, reduced competition and predation threats, and access to artificial shelter (Bateman and Fleming 2012; Šálek et al., 2015; Widdows and Downs 2016). The benefits derived from these can increase localised population densities and adaptions in species behavioural ecology (Bateman and Fleming 2012). Consequently, various aspects of urban carnivore ecology have been investigated as conservation of these species in the urban landscape relies on understanding their spatial ecology (e.g. home range size, composition, distribution and habitat use).

Mongooses are widely distributed throughout Africa, occupying a range of diverse habitat types (Jennings and Veron 2020; Skinner and Chimimba 2005). Members of the Herpestidae taxa are noted for their opportunistic diet and behavioural plasticity (Jennings and Veron 2020; Skinner and Chimimba 2005). The behavioural characteristics that members of this taxon share increase their likelihood of becoming a successful African synanthropic species in the urban environment (Admasu et al., 2004; Jennings and Veron 2020; Skinner and Chimimba 2005). Mongoose species have the potential to become habituated to human activity and exploit urban landscapes which could result in changes to the species population ecology, behaviour, morphology, and physiology. However, the general ecological knowledge of the Herpestidae taxa remains limited and outdated in Africa (Rood 1986; Streicher et al., 2020).

Furthermore, this taxon's spatial urban ecology remains understudied although they are widely distributed in urban areas (Skinner and Chimimba 2005). The effects of urbanisation can provide insight into a particular taxon's ability to adapt to increasing anthropogenic pressures. These innate characteristic places mongooses as a potentially appropriate model for how African mesocarnivores can persist and adapt to an urban landscape and its associated constraints. As little conservation is presently focussed on the various Herpestidae taxa, this research will allow us to create and align conservation measures to ensure their protection.

We investigated the spatial ecology of water mongoose (*Atilax paludinosus*) within a fragmented urban matrix. Water mongoose is widely distributed across southern Africa (Skinner and Chimimba 2005). The species is solitary; however, they have been reported to forage in family groups (Skinner & Chimimba, 2005). The water mongoose is restricted to riparian habitats with dense vegetation cover, and estuaries in coastal areas (Baker and Ray 2013; Do Linh San et al., 2015; Skinner and Chimimba 2005). To gain insight into the urban ecology of water mongoose, we determined the home range and habitat use in the Upper Highway Area's urban matrix, Durban, KwaZulu-Natal Province, South Africa, using Global

Positioning System (GPS)-Ultra high frequency (UHF) transmitter telemetry. We hypothesised that home range sizes for water mongoose species would differ from previous studies conducted in natural/semi-natural habitats. We predicted that the species' home range sizes would be reduced within the urban matrix landscape compared with those in natural or farmland landscapes determined in previous studies.

## 4.3 Methods

*Study area*

Our study was conducted in selected residential suburbs of the Upper Highway Area of eThekwini Municipality, Durban, KwaZulu-Natal Province, South Africa (Figure 4.1). The Upper Highway Area or western suburb of eThekwini includes the residential areas of Kloof and Hillcrest and smaller areas of Assagay, Botha's Hill, Forest Hills, Gillitts, Waterfall and Winston Park (Smith 2017). The study area is a mosaic of mixed levels of urban developments (commercial, industrial or residential use) interspersed by green spaces (residential gardens, parks, conservancies, green belts and nature reserves) (Municipality 2007; Roberts 1994). Natural vegetation persists in conservancies and nature reserves under the Durban Metropolitan Open Space System (D'MOSS) policy within this urban matrix system of green spaces and corridors that link essential biodiversity conservation sites in eThekwini Municipality to facilitate passage of fauna and flora (Roberts 1994). The Upper Highway Area experiences mean annual minimum and maximum temperatures of 13.9°C and 24.0°C, respectively (http://en.climate-data.org/location/27097/). The area receives most of its 974 mm mean annual rainfall during the summer months (December-February) (Mucina and Rutherford 2006). The study site's natural vegetation consists of KwaZulu-Natal Sandstone Sourveld grassland with patches of riverine forest and scarp forest (Mucina and Rutherford 2006). The area has

undergone significant anthropogenic transformations for agriculture and urban infrastructure (buildings, roads and human settlements) (Mucina and Rutherford 2006; Municipality 2015).

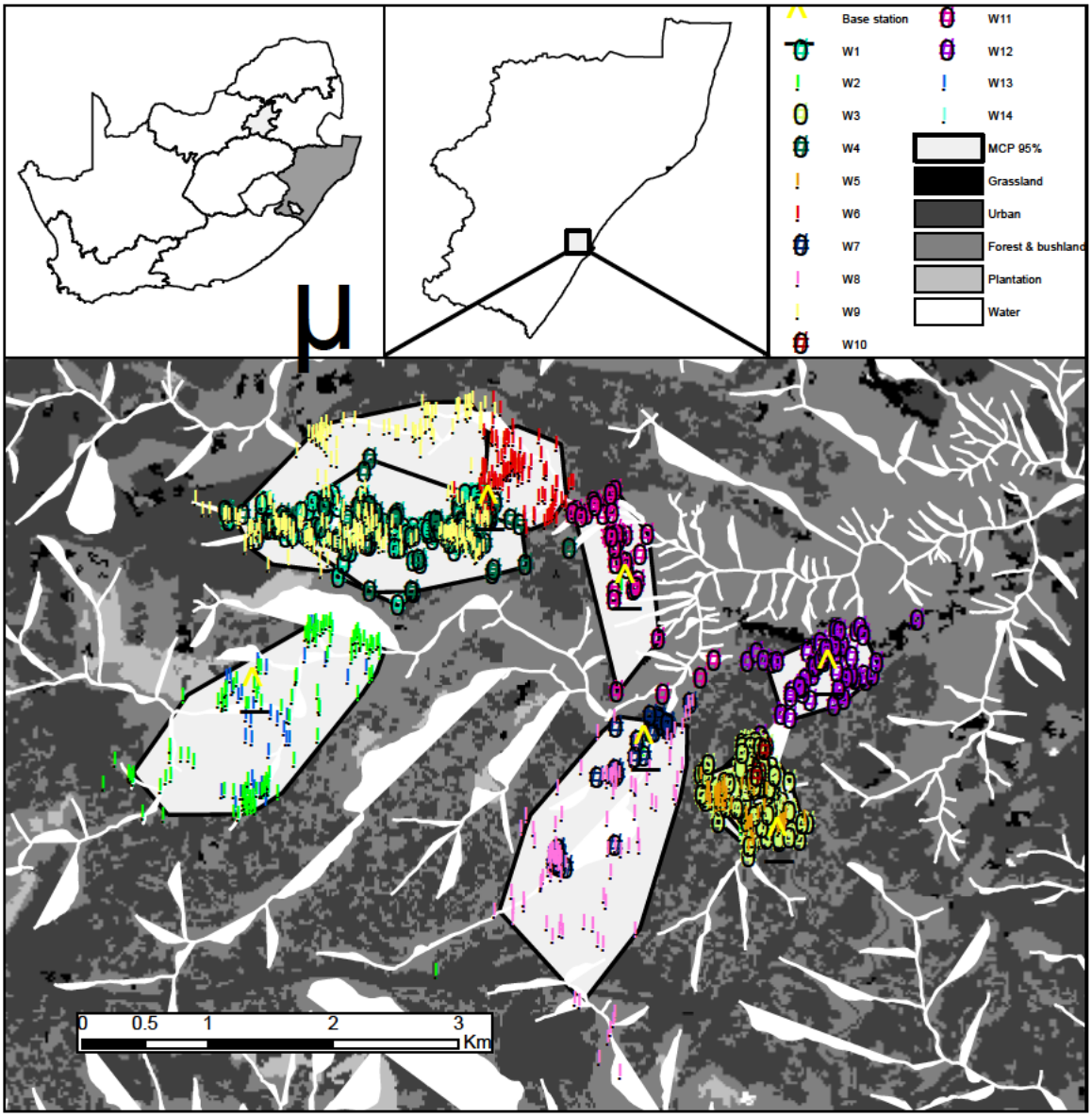


**Figure 4.1.** Space use of water mongoose (triangles represent female and circles male) in the urban mosaic in the present study in the Upper Highway Area of eThekwini Municipality, Durban, KwaZulu-Natal Province, South Africa.

*Data collection*

We captured and collared water mongoose between July 2018 and May 2019. We selected trap locations based on areas frequented by the water mongooses, which were further

attracted with bait. We used five steel, step plate cage traps dimensions (50 x 50 x 100 cm) were rotationally set in dense vegetative areas in different sites (grasslands, dense bush, streams and forest) in the study area and baited with chicken hearts and chicken intestines. We camouflaged the traps with indigenous vegetation (grasses, branches and surrounding trees) because of a high failure rate associated with uncovered traps (Maddock 1988; Streicher et al., 2020). Traps were set continuously for 24 h periods during which they were checked following ethical procedures (three times per day: morning, early afternoon, and evening). We calculated trap success using the number of individuals captured divided by the total effort (number of trapping days x number of traps), multiplied by 100.

With veterinarians', we sedated, and fitted GPS/UHF collars on the trapped individual of water mongooses (University of KwaZulu-Natal Animal Ethical clearance No. 020/15/animal). Captured mongooses were shifted to a containment trap and sedated by an intramuscular injection on-site by a veterinarian based on an estimated weight. Sedation drugs – Anaket® (ketamine) (0.8 mg/kg, Bayer, South Africa) and Domitor® (medetomidine) (0.5 mg/kg, Pfizer, South Africa) were used to immobilise mongooses. Once immobilised, mongooses were sexed, and morphometric measurements and blood samples were taken. Individual mongooses were fitted with a GPS-GSM/UHF neck collar when the total collar mass was less than 3% of the animal's body mass (a minimum of 1.75 kg) (Animal Trackem, Pietermaritzburg, South Africa) (Boitani and Fuller 2000; Kenward 2001). A finger width gap was left between the collar and the neck of collared mongooses to allow for additional growth and prevent discomfort for the animal. Successfully collared mongooses were given a reversal anaesthetic of Antisedan® (atipamezole) (0.5 mg/kg, Zoetis, South Africa). To avoid autocorrelation and ensure statistical independence of GPS points, each collars' GPS fix rate was set to record at 4 h intervals from 18:00–06:00. We set the time intervals of telemeters following the study species' natural activity patterns as found in the literature. We set up an

array of four portable solar-powered base–station receivers (~ 5 km radius) across the study area to download telemetry data with a spatial accuracy of ~ 5 m from collared mongoose when they were in range. Base–station receivers received telemeter data then sent these via cellular network (GSM), and we accessed data from the Wireless Wildlife webpage (https://www.wireless-wildlife.co.za/index.html).

*Home range analyses*

We downloaded and formatted GPS data from each mongoose in Microsoft Excel. We then screened and filtered the GPS data using R packages adehabitatLT' *version 0.3.20* (Calenge 2006), *'adehabitatMA' version 0.3.10* (Calenge 2006), *'ade4' version 1.7e4* (Dray et al., 2015) and *'sp' version 1.2e3* (Pebesma and Bivand 2005) in R software (version 3.1.2) (RStudio 2015) to remove erroneous data points (Laver and Kelly 2008; Laver et al., 2015). Site fidelity was assessed for each individual before home range analyses (Signer and Balkenhol 2015). Two measures of the centre of activity for site fidelity were used namely the Mean Square Distance (MSD) and Linearity index (LI) in 'rhr' version 1.2.909 package in R (Signer and Balkenhol 2015). MSD is the level of dispersion use measured around the home range centroid, whereas LI is the linear distance between endpoints of an animal traversed movements divided by the total distance (Bell and Kramer 1979; Schoener 1981). MSD and LI actual values are compared to the mean and standard errors generated from random animal trajectories for each individual (Spencer et al., 1990). An individual is judged to show site fidelity when the actual MSD and LI values are significantly less ($p < 0.05$) than the randomly projected mean values for both measures (Spencer et al., 1990). We calculated home range estimates using the Reproducible Home Range (RHR) GUI package in R following the criteria set by Laver and Kelly (2008). Multiple home range estimates were used in the study because of the different method advantages and disadvantages of each estimate (Laver and Kelly 2008).

Additionally, the incorporation of multiple methods allows comparison to be drawn been past studies because of a lack of unified methodology for estimating animal home ranges among researchers (Börger et al., 2020; Laver and Kelly 2008). The three home range estimates were, Maximum Convex Polygon (MCP), Kernel Density Estimate (KDE) and Local Convex Hull (LoCoH) utilised at the 95% and 50% core area limits. Buffers and resolutions for each mongoose were manually determined for the KDE home range estimate using the $h_{plug\text{-}in}$ bandwidth. The $h_{plug\text{-}in}$ bandwidth is the recommended bandwidth when analysing KDE in fragmented, patchy urban environments within small geographic zones because it is more conservative than $h_{lscv}$ which results in less smoothing than $h_{lscv}$ (Amstrup et al., 2004; Girard et al., 2002; Gitzen et al., 2006; Walter et al., 2011). We calculated the home range for each mongoose from a minimum of 30 GPS location during the tracking duration. We used the Mann-Whitney *U*-test to estimate any significant difference between home range size between 1: male and female and 2: sub-adult and adult water mongooses. Age class was estimated based on body size, reproductive condition and tooth wear (Ramesh et al., 2015). We conducted a repeated measure analysis of variance (RMANOVA) to compare the three home range estimation methods for water mongoose in Statistica (Statsoft Inc., Tulsa, OK, USA).

*Habitat use*

We investigated the relationship between water mongoose habitat use from the GPS data and habitat variables. A land-use map with a 20 m resolution from Ezemvelo KZN Wildlife (EKZNW) was used (Wildlife and GeoTerraImage 2018). We reclassified the land-use categories into six broad land-use types (grassland, forest and bushland, water, urban built and exotic timber plantation). Second-order habitat use was estimated using the pooled 100% MCP home range of water mongoose in the study site, which was clipped to the reclassified land-use types. Third-order habitat use was estimated using the 100% MCP home ranges for each mongoose. The available habitat type proportions in the respective mongoose species

individuals home ranges were determined using ArcGIS 10.6 (ESRI, Redlands, CA, USA). Furthermore, we overlaid GPS points on the reclassified land-use map to determine habitat types for the observed locations.

We assessed individual mongoose habitat use as the observed proportion of location in the land use types (Gehrt et al., 2009). Selection of habitat types was calculated using the null model of the differences between the observed compared with the expected based on availability (Byers et al., 1984; Neu et al., 1974). Habitat use was calculated using the 2$^{nd}$ and 3$^{rd}$ order for water mongoose (Gehrt et al., 2009; Johnson 1980). Chi-squared Goodness of fit test was performed for observed vs expected frequencies of habitat types to determine significance in habitat type used at the 2$^{nd}$ (individual home range scale) and 3$^{rd}$ (landscape scale) order for the water mongoose (Byers et al., 1984; Johnson 1980; Neu et al., 1974). A Bonferroni post-hoc test was performed to establish a significant difference between the use of habitat types (Byers et al., 1984). Habitat preference was, therefore inferred if the expected use range was above or below the Bonferroni confidence interval for each habitat type (Byers et al., 1984). Habitat types that were not used by individuals were allotted a small value (0.001) to avoid division by zero (Aebischer et al., 1993; Humphries et al., 2016).

*Spatial overlap*

In this study, we used the static analysis outlined in (Kernohan et al., 2001) to estimate spatial overlaps between and within sexes of urban water mongooses. The pairwise overlap analysis where $HR_{ij}$ is the proportion of an individual's (*i*) home range that overlaps with another individual's home range (*j*), which allows the shared spatial area $A_{ij}$ to be calculated form the two individuals home ranges ($A_i$ and $A_j$) (Fieberg and Kochanny 2005; Kernohan et al., 2001). Home range measures of MCP were used to calculate the average proportion of home range overlap at both the core area 50% and 95% level between and within sexes and age classes. Not all water mongooses were tracked during the same period; therefore the spatial

overlaps analysis represents only an estimate and should be treated with caution (Carter et al., 2012).

**Table 4.1** Details (species, sex, age class start date, end date, number of GPS fixes, duration and body mass) of individuals of water mongoose collared with GPS/UHF transmitters in an urban mosaic landscape in the present study.

| Individual | Species | Sex | Age class | Mass (kg) | Start Date | End Date | No. of GPS fixes | No. of Days |
|---|---|---|---|---|---|---|---|---|
| W1 | Water | F | Adult | 3.42 | 14/07/2018 | 03/11/2018 | 493 | 112 |
| W2 | Water | M | Adult | 3.40 | 16/07/2018 | 19/12/2018 | 201 | 156 |
| W3 | Water | F | Adult | 3.05 | 17/07/2018 | 07/02/2019 | 325 | 205 |
| W4 | Water | F | Sub-adult | 2.20 | 17/07/2018 | 12/09/2018 | 144 | 57 |
| W5 | Water | M | Sub-adult | 2.52 | 18/07/2018 | 29/07/2018 | 92 | 11 |
| W6 | Water | M | Adult | 3.06 | 22/07/2018 | 28/12/2018 | 164 | 159 |
| W7 | Water | F | Sub-adult | 2.82 | 23/07/2018 | 18/08/2018 | 50 | 26 |
| W8 | Water | M | Adult | 3.60 | 26/07/2018 | 03/02/2019 | 186 | 192 |
| W9 | Water | M | Adult | 3.42 | 26/07/2018 | 06/01/2019 | 345 | 164 |
| W10 | Water | F | Sub-adult | 2.90 | 24/08/2018 | 26/08/2018 | 2 | 2 |
| W11 | Water | M | Adult | 3.10 | 26/05/2019 | 04/10/2019 | 45 | 131 |
| W12 | Water | F | Adult | 3.50 | 28/04/2019 | 18/10/2019 | 87 | 173 |
| W13 | Water | F | Adult | 2.90 | 28/04/2019 | 18/10/2019 | 88 | 173 |
| W14 | Water | M | Adult | 3.00 | 05/05/2019 | 07/06/2019 | 7 | 33 |

## 4.4 Results

We captured a total of 22 water mongoose of which we radio-collared 14 with a trap success rate of 7.2% for our study. Mesocarnivore bycatch species included cape genet (*Genetta tigrina*) (n =13), large grey mongoose (*Herpestes ichneumon*) (n = 2), domestic dog (*Canis lupus familiaris*) (n = 3) and domestic cat (*Felis catus*) (n = 2). Water mongooses were radio-tracked in an equal ratio sex ratio (Table 4.1). The mean body mass of collared water mongooses was 3.06 ± 0.39 kg. Male water mongooses (3.16 ± 0.36 kg) were on average heavier than female water mongooses (2.97 ± 0.43 kg), however the difference was not significant (independent samples T-test, t = -0.884, df = 12, p = 0.394).

*Home range estimates*

We obtained a total of 2229 GPS locations from tagged mongooses. The study site's interference caused by landscape features (topographical, vegetation and buildings) hindered GPS fixes being generated by the tracking units. GPS fix rates ranged between 4% - 50% per individual for a time duration ranging between 2 - 205 days (Table 4.1). Individuals W5, W10 and W14 collars ceased after 11, 2 and 33 days (Table 4.1). It was not possible to run home range analyses on individuals W10 and W14 because of collar failure and an inadequate number of GPS fixes. The MCP, KDE and LoCoH methods provided different results (Figure 4.2). The home range for water mongoose showed individual variation (Figure 4.2a). Water mongoose mean (± S.E.) 95% confidence home range varied between the three home range estimates MCP = 1.0 ± 0.19 $km^2$, KDE = 0.4 ± 0.08 $km^2$ and LoCoH = 0.4 ± 0.09 $km^2$ (Figure 4.2b). On average, male water mongoose had a larger home range than female water mongoose for all three home range estimates (Figure 4.2c). Home range 95% estimates did not differ significantly between the sexes of water mongoose (Mann-Whitney U-test; MCP95%, $U$ = 13.0, p = 0.423, KDE95%, $U$ = 15.5, p = 0.700, and LoCoH95 $U$ = 10.0, p = 0.200). On

average, sub-adult water mongooses had a larger home range than adult water mongoose for MCP and KDE estimates for home range (Figure 4.3a). Home range 95% estimates did not differ significantly between the age classes (Mann-Whitney U-test; MCP95%, $U$ = 16.0, p = 1.000, KDE95%, $U$ = 11.0, p = 0.395, and LoCoH95 $U$ = 16.0, p = 1.000)

The home range estimators used differed significantly for water mongoose for the 95% home range (RMANOVA, $F_{(2, 22)} = 8.001$, p = 0.003, Supplementary information Figure S4.1). The Bonferroni post hoc test indicated significant differences in the home range estimates when MCP home range results were compared with KDE (P = 0.008) and LoCoH (P = 0.005) home range measures.

The mean 50% core range for water mongoose varied between the three home range estimates MCP = 0.38 ± 0.11 $km^2$, KDE = 0.05 ± 0.01 $km^2$ and LoCoH = 0.2 ± 0.01 $km^2$. Male water mongoose on average had larger core home ranges than female water mongoose across all three estimates (Figure 4.2d). Core home range 50% estimates did not differ significantly between the sexes of water mongoose (Mann-Whitney U-test; MCP50%, $U$ = 14.5, p = 0.580, KDE50%, $U$ = 13.5, p = 0.470, and LoCoH50 $U$ = 15.0, p = 0.631). Sub-adult water mongooses on average had a larger core area use (Figure 4.2c) than adult water mongoose for all estimates for core area (Figure 4.3b). Core area 50% estimates did not differ significantly between the age classes for water mongoose (Mann-Whitney U-test; MCP50%, $U$ = 15.0, p = 0.865, KDE50%, $U$ = 14.5, p = 0.799, and LoCoH50 $U$ = 12.0, p = 0.497).

The home range estimators differ significantly for water mongoose for the 50% core area use (RMANOVA, $F_{(2, 22)} = 7.291$, p = 0.004, Supplementary information Figure S4.2). The Bonferroni post hoc test indicated significant differences in the home range estimates when MCP core area use results were compared with KDE (P = 0.015) and LoCoH (P = 0.007).

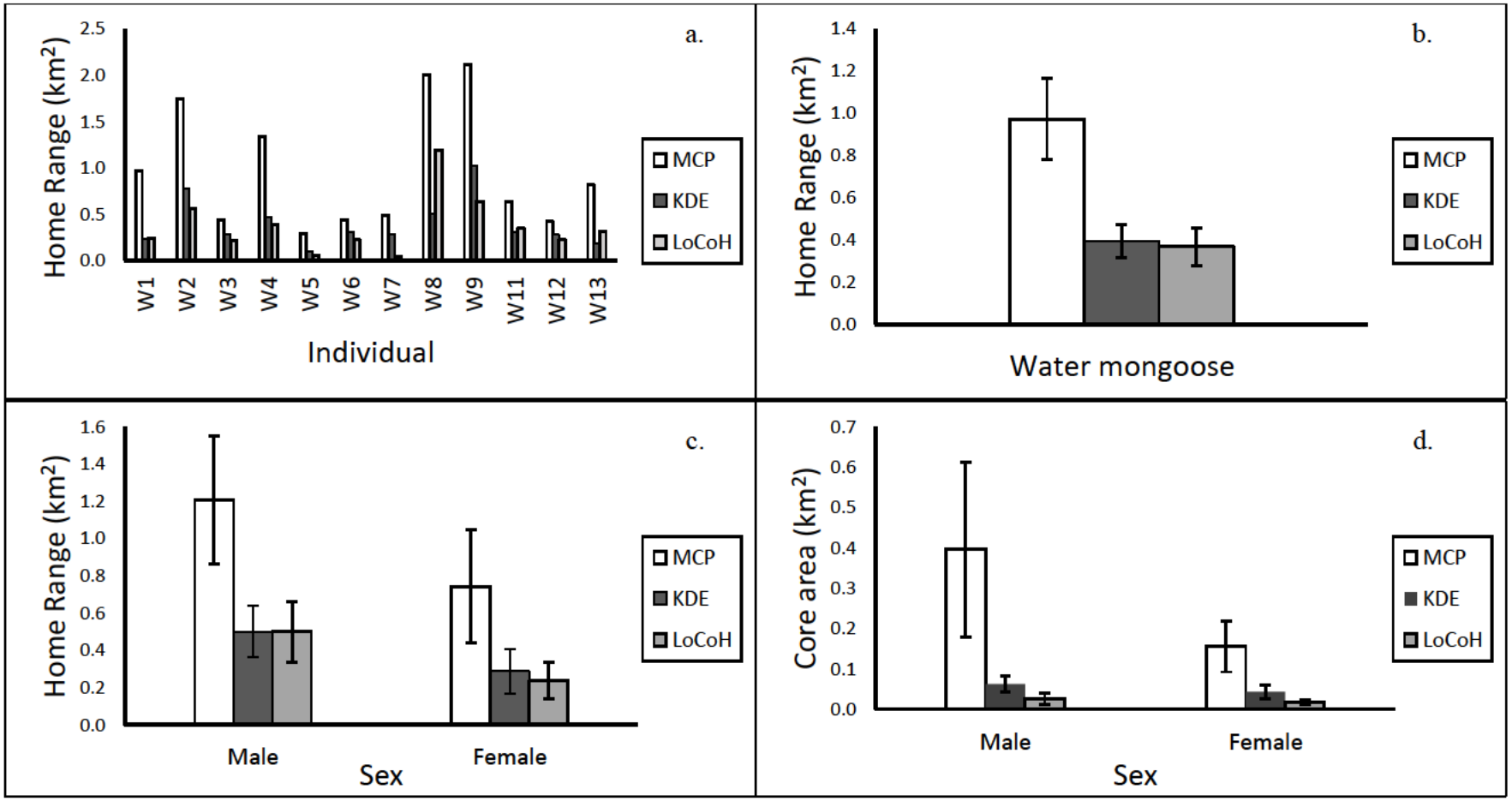


**Figure 4.2.** a. Individual home ranges of water mongoose (W, n = 12), b. overall mean home range (± S.E.) size for water mongoose and c. mean home range between sexes for water mongoose using the 95% home range (± S.E.) and d. mean core area size between sexes for water mongoose using the 50% core area of the three home range estimates (maximum convex polygon, kernel density estimate and local convex hull) in the present study.

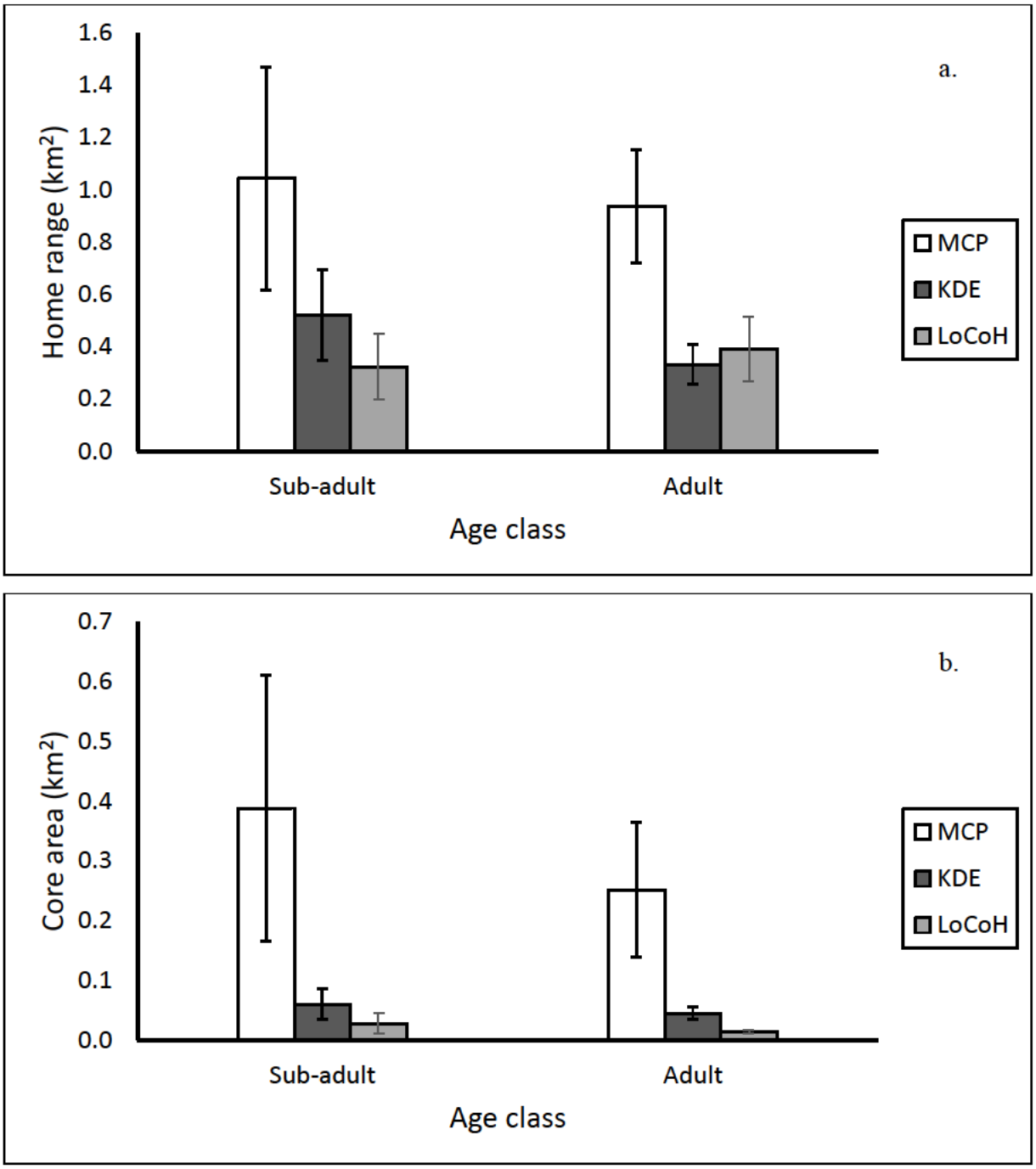


**Figure 4.3.** The mean home range between age class for water mongoose using the 95% home range (± S.E.) and b. mean core area size between age class water mongoose using the 50% core area of the three home range estimates (maximum convex polygon, kernel density estimate and local convex hull) in the present study.

*Habitat use*

*Landscape-scale (2nd order)*

Second-order habitat preference differed from that of the study area for water mongoose. Forest and bushlands, water and exotic timber plantation land-use types were used more than expected based on their availability throughout the study area for water mongoose. Urban built, and grassland land-use types were avoided based on availability (Table 4.2).

*Home range scale (3rd order)*

At the finer scale, water mongoose's habitat use was not uniform within an individual's home ranges (Table 4.3 and Supplementary information Table S4.1). There was individual variation present within 3rd order habitat use of 100% MCP for water mongooses (Table 4.3). There was a mixed level of use of different land-use types in the urban mosaic landscape; however, all water mongoose avoided urban built land-use type (Table 4.3). Water mongooses selected or used in portion, forest and bushland and water land-use features, except for water mongoose W5, W7 and W13 who avoided water land-use type. Variation in habitat use was observed between 2nd and 3rd order habitat selection for all habitat types except urban built areas that were avoided across the orders (Table 4.2, Table 4.3 and Supplementary information Table S4.1).

*Spatial overlap*

The degree of spatial overlap between collared water mongooses varied within and between sexes (95% mean MCP overlap: female-female 81.0 ± 18.38%, male-male 2.5 ± 2.12%; female–male 60.3 ± 42.15% and male-female 37.3 ± 32.34%) (Figure 4.4). There was no significant relationship present in which the degree of 95% home range overlap occurred within and between the sexes ($F = 2.206$, $df = 3$, $p = 0.133$). Degree of intraspecific core area overlaps in water mongooses were lower

than home range overlap with overlaps (50% mean MCP overlap: female-female 75.0 ± 29.70%, female–male 59.8 ± 35.27% and male-female 24.8 ± 17.86% Figure 4.4). Additionally, there was no core area overlaps present between male water mongooses. There was no significant relationship present in which the degree of 50% core area range overlap occurred within and between the sexes ($F = 2.617$, $df = 2$, $p = 0.142$).

Spatial home range overlap between and within age classes showed variation (95% mean MCP overlap: adult – adult 34.3 ± 13.98%, adult – sub-adult 47.0 ± 15.34%; sub-adult – adult 68.2 ± 39.54% (Figure 4.5). There was no significant relationship present in which the degree of 95% home range overlap occurred within and between age classes ($F = 1.215$, $df = 2$, $p = 0.324$). Overlap in the core area range was higher between adults – adults and adults – sub-adults compared to home range overlap (50% mean MCP overlap: adult – adult 48.3 ± 20.87, adult – sub-adult 55.3 ± 22.33%; sub-adult – adult 43.0 ± 14.64%) (Figure 4.5). There was no significant relationship present in which the degree of 50% core area range overlap occurred within and between the sexes ($F = 0.085$, $df = 2$, $p = 0.920$). There was no spatial overlap present between sub-adults for both the home range and core area range measures.

**Table 4.2** Habitat availability within the 3rd order pooled 100% maximum convex polygons (MCP) in the urban mosaic landscape of Kloof study area based on the Bonferroni confidence intervals of water mongoose habitat selection.

| Habitat type | Contribution (%) | Area ($km^2$) | $P_{io}$ | $P_i$ | Bonferroni CI | Conclusion |
|---|---|---|---|---|---|---|
| Grassland | 0.98 | 0.21 | 0.0098 | 0.0004 | -0.001 < P < 0.002* | Avoidance |
| Forest & bushland | 48.09 | 10.21 | 0.4809 | 0.6491 | 0.623 < P < 0.675* | Selection |
| Water | 3.66 | 0.78 | 0.0366 | 0.1345 | 0.116 < P < 0.153* | Selection |
| Urban built | 44.63 | 9.48 | 0.4463 | 0.1772 | 0.156 < P < 0.198* | Avoidance |

| | | | | | | |
|---|---|---|---|---|---|---|
| Exotic timber plantation | 2.64 | 0.56 | 0.0264 | 0.0387 | 0.028 < P < 0.049* | Selection |

$P_{io}$, Expected proportion of usage; $P_i$, Actual proportion of usage; Bonferroni CI, Bonferroni confidence intervals; *, a significant difference at $P < 0.05$.

**Table 4.3** Habitat use of water mongoose using Bonferroni confidence intervals with the 100% maximum convex polygon (MCP) home range in the present study.

| Mongoose ID | Species | Habitat types | | | | |
|---|---|---|---|---|---|---|
| | | Grassland | Forest & Bushland | Water | Urban built | Exotic timber plantation |
| W1 | Water | - | + | + | - | - |
| W2 | | - | + | ± | - | ± |
| W3 | | ± | + | + | - | NIL |
| W4 | | NIL | + | + | - | - |
| W5 | | - | + | - | - | NIL |
| W6 | | - | ± | + | - | NIL |
| W7 | | NIL | + | - | - | NIL |
| W8 | | NIL | + | ± | - | NIL |
| W9 | | - | + | + | - | ± |
| W10 | | NIL | NIL | NIL | NIL | NIL |
| W11 | | NIL | NIL | NIL | NIL | NIL |
| W12 | | - | ± | + | - | NIL |
| W13 | | ± | ± | - | - | ± |
| W14 | | NIL | NIL | NIL | NIL | NIL |

*- Indicates avoidance, + indicates a preference, ± indicates utilised in proportion to availability

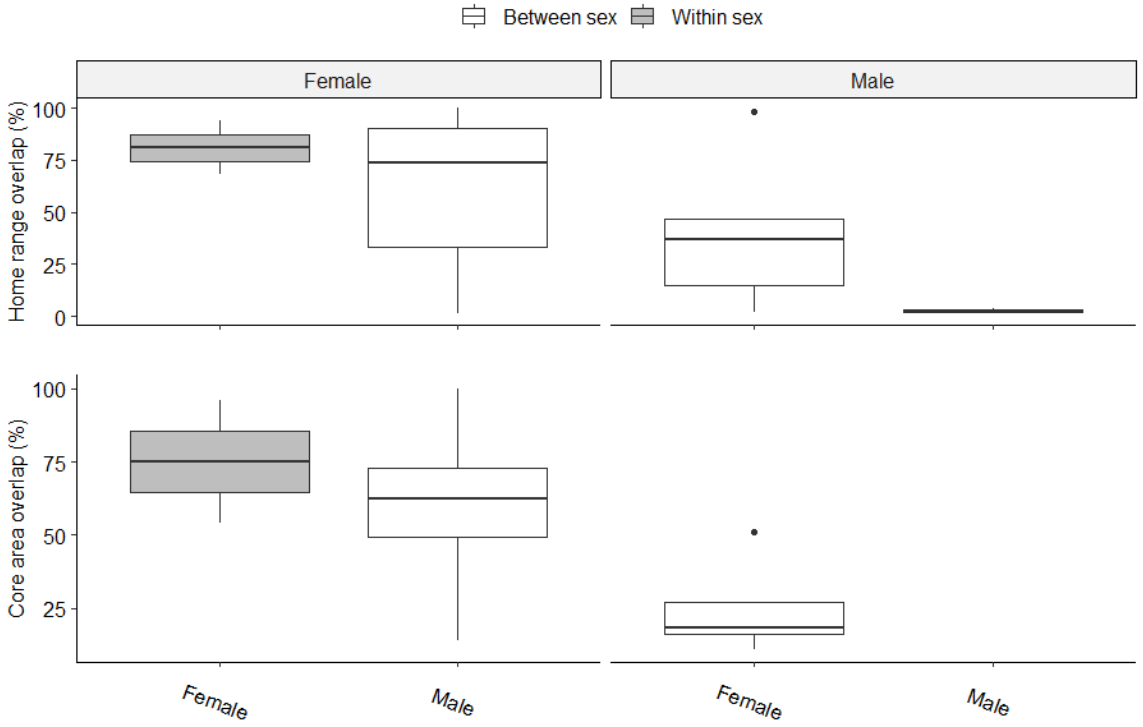


**Figure 4.4.** The home range and core area overlap between and within sexes of water mongoose using MCP home range estimate of collared mongooses in the present study.

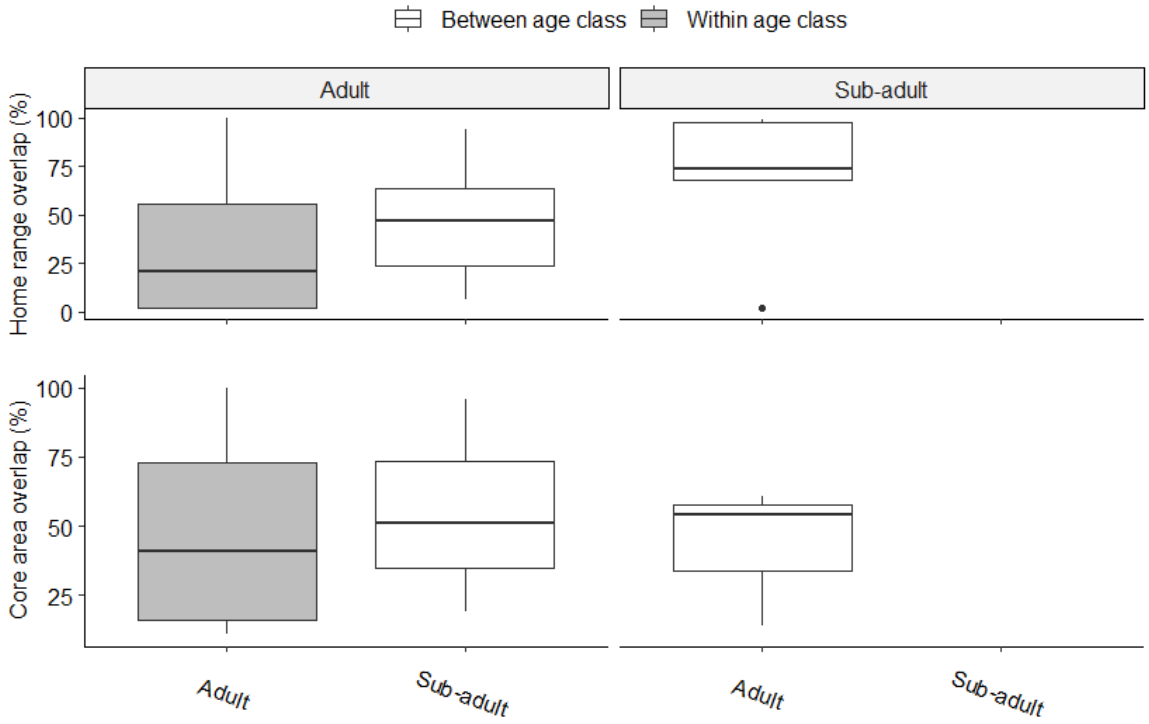


**Figure 4.5.** The home range and core area overlap between and within age classes (Adult and Sub-adult) of water mongoose using MCP home range estimate of collared mongooses in the present study.

## 4.5 Discussion

The process of urbanisation of previously natural areas has led to significant natural habitat loss for water mongooses. The study emphasises how water mongooses can adapt to the anthropogenic pressures of urbanisation. The species can persist in the Durban Upper Highway Area's semi-urban habitat by reducing individual home range size and utilising natural habitat spaces. However, water mongooses were mainly restricted to fragmented natural patches (D'MOSS), conservancies, or Krantzkloof Nature Reserve within the residential urban mosaic. The restriction of water mongooses spatial habits to dense vegetation cover and the study sites' topographical nature may have attributed to why our study experienced high failure rates in GPS fixes (pers. obs.).

Our study further illustrates the inter-individual variation in both home range and habitat use for water mongooses in this urban mosaic. Differences in physiological and behavioural traits in conjunction with abiotic conditions (landscape type, food availability, level of fragmentation, anthropogenic pressures) are plausible explanations for this individual variation (Bateman and Fleming 2012; Gehrt et al., 2010; Newsome et al., 2015). The retention of these urban natural and/managed green spaces (D'MOSS in this study) is critical for sustaining urban wildlife (Alexander et al., 2019; McPherson et al., 2019; Zungu et al., 2020). Several mesocarnivores referred to as "urban exploiters" (dingo: *Canis familiaris dingo*, caracal: *Caracal caracal*, and bobcat: *Lynx rufus*), have been shown to avoid built urban areas strictly, and exclusively use these green spaces in the urban landscape matrix. (Leighton et al., 2020; McKinney 2006; McNeill et al., 2016; Riley et al., 2003).

Home range estimation is crucial to understanding how animals perceive and use their spatial environment (Akcakaya 2000; Thomas and Taylor 2006). We estimated home ranges and core area use of water mongooses using three different estimation methods MCP, $KDE_{plug\text{-}in}$ and LoCoH. The three different estimation methods produced different home range and core area sizes. MCP had markedly more extensive home range and core area sizes than KDE and LoCoH estimates in this study. MCP is a crude estimation method sensitive to outlier data points and sample size, whilst implying uniform use

of data points that fall within its boundary (Powell 2000). A minimal difference was present between $KDE_{plug\text{-}in}$ and LoCoH measures, which are conservative. In particular, LoCoH hard definitive boundaries can underestimate home range size (Huck et al., 2008). Given this, we support previous works that advocate for multiple home range estimation approach (Chibesa et al., 2017; Pebsworth et al., 2012; Reinecke et al., 2014).

The home range sizes for urban water mongooses were significantly smaller than home range sizes and core area use than GPS telemetry results for water mongooses occurring in the fragmented natural habitat and farmland mosaic of the KwaZulu-Natal Midlands (Streicher et al., 2020). The mean 95% MCP and 95% LoCoH home range sizes for water mongoose from Streicher et al., (2020) (mean 95% MCP: 11.8 ± 1.52 $km^2$, range 6.80–16.33 $km^2$, 95% LoCoH: 4.6 ± 0.78 $km^2$, range 1.47–10.17 $km^2$) were significantly larger than the present urban study (Figure 4.2). Similarly, overall core area use measures of water mongooses (Streicher et al., 2020); mean 50% MCP: 2.2 ± 0.86 $km^2$, range 0.76–4.45 $km^2$, 50% LoCoH: 0.3 ± 0.12 $km^2$, range 0.04–0.7 $km^2$) were greater than those core area use estimates in our urban study. We could not compare KDE values because different bandwidth measures ($h_{ref}$ vs $h_{plug\text{-}in}$) were used in the two studies. The reduction in water mongoose home range size in the urban mosaic was as predicted in the present study. Generally, mammalian mesocarnivores have been recorded to alter their behaviour in response to anthropogenic transformed landscapes. We were unable to compare KDE values because different bandwidth measures ($h_{ref}$ vs $h_{plug\text{-}in}$) were used in the two studies. The reduction in home range sizes of water mongooses was predicted in the present study. Generally, mammalian mesocarnivores have been recorded to alter their behaviour in response to anthropogenic transformed landscapes (Bateman and Fleming 2012; Gehrt et al., 2010; Šálek et al., 2015; Widdows and Downs 2016). Their behavioural plasticity enables mesocarnivores to adapt to, and persist in, an urban mosaic landscape. Core area use indicated areas of high habitat use by water mongooses. These high use areas were in dense vegetation close to water sources in the green spaces of D'MOSS and nature reserves in the study area.

Our results showed that habitat selection by water mongooses varied with scale and was individual-specific. At the landscape scale and home range scale, water mongooses selected for or used in proportion, only forest and bushland areas with other habitats having variable levels of usage. However, built urban areas were avoided entirely at both the landscape and home range scale. Therefore, we can conclude that water mongoose at both scales selected for forest and bushland habitat (green spaces in the urban landscape matrix). This highlighted the importance of forest and bushland vegetation structure in the D'MOSS areas for water mongoose. Complete avoidance of built urban areas at both scales illustrated the negative effect this land-use type has on this species. It must be emphasised that the use of available landscape types within an individual's home range is limited by their territorial behaviour, particularly for male water mongooses. The D'MOSS green space mosaic provides crucial habitat niches that allow water mongooses to persist in this urban matrix, further emphasising its importance in maintaining and promoting biodiversity within the urban mosaic of Durban (Maseko et al., 2020; McPherson et al., 2019; Zungu et al., 2020).

Water mongoose's spatial response was concordant with trends illustrated for other urban mesocarnivores (Eurasian badger; *Meles meles*, red fox: *Vulpes vulpes*, striped skunk: *Mephitis mephitis*, raccoon: *Procyon lotor* and cape genet). This illustrated that urban mesocarnivores generally had reduced home range sizes than non-urban conspecifics (Gehrt et al., 2009; Gehrt et al., 2010; Šálek et al., 2015; Widdows 2016). Reductions in mesocarnivore home range sizes in urban areas can be because of various factors including concentrated food resources, fragmented suitable habitats, access to den sites, and anthropogenic boundaries (roads, walls and canals etc.) that impede the free movement of animals (Bateman and Fleming 2012). Food resources are suggested to be more concentrated and abundant in urban areas; therefore, urban mesocarnivores can obtain the necessary resources from a smaller area (Gehrt et al., 2010; McKinney 2006). Alternatively, mesocarnivores that avoid urban built areas are confined to the fragmented suitable habitat "green spaces" in the urban mosaic to obtain food resources. The level of development and connectivity between "green patches" are suggested to affect

mesocarnivores' movement ecology. The D'MOSS patches and corridors typically can act as pockets of refugia for mesocarnivores and other species in built-up areas (Alexander et al., 2019; Maseko et al., 2020; McPherson et al., 2019; Zungu et al., 2020). However, these green spaces' "refugia" are potential traps in urban areas that lack connectivity between other green patches (Lepczyk et al., 2017). Anthropogenic boundaries and stresses could be preventing the free movement of mongooses in the area. Thus, conservation should target efforts to connect isolated refugia. Increasing connectivity between isolated refugia would improve dispersal for mongooses, thus promoting genetic transfers and possibly population growth and natal range dispersal.

Our results illustrated that the 'forest and bushland' was the preferred habitat over other natural habitats in the study area at both the landscape and home range scale for water mongooses. These 'forest and bushland' areas are beneficial for the species for several reasons. Firstly, these green spaces are protected by D'MOSS initiatives that conserve natural areas for animals and human recreation in the urban matrix of the Durban Metropole. These green spaces have reduced human disturbance and stresses compared with built-up regions; therefore, water mongoose can find refuge in these green spaces. Secondly, green spaces reduce interactions with domestic pets. However, there is a lack of data that mesocarnivores alter their natural movements in response to domestic pets and associated fatality risks (Gehrt et al. 2010). Widdows et al. (2018) showed that domestic dogs were the primary cause of fatalities of cape genets in the greater Durban area. This particular threat faced by mongooses in the study area was emphasised with five uncollared water mongooses reportedly killed by domestic dogs on landowners properties during the present study. All these water mongooses were sub-adults (J.S. pers. obs.), suggesting that the species faces a substantial threat from domestic pet interactions when attempting to disperse from their natal range. Lastly, these green spaces are enclosed and transacted by an active road network system. The M33, a major motorway in the area, cuts through the Krantzkloof Nature Reserve and runs parallel to several patches of D'MOSS and conservancies in the

area. This motorway and other road networks are a significant threat to wildlife (Widdoes et al. 2018), particularly when terrestrial animals attempt to disperse.

There were differing degrees in which collared water mongooses overlapped within and between sexes for both the 50% core area unitisation and 95% home range in this study. Female water mongoose core area use and home range had high spatial overlap with other female and male water mongooses. Female water mongooses are typically non-territorial towards conspecifics (Skinner and Chimimba 2005). The high spatial overlap is further emphasised when sub-adults fail to disperse from natal philopatry, which can result in the formation of small conspecific groups. (Skinner and Chimimba 2005; Streicher et al., 2020).

Interestingly, male water mongooses had comparatively lower levels of spatial overlap with females at both the core area unitisation and home range levels. Despite the reduced home range size of water mongooses in our urban study, there was no overlapping for core use and minimal overlapping home range present between male water mongooses. Male water mongooses are suggested to be highly territorial towards other males, with their spatial range only overlapping with potential mates (Skinner and Chimimba 2005; Streicher et al., 2020).

The present study highlighted water mongoose ability to persist in green refugia spaces on the fringe of an urban matrix landscape. The avoidance of anthropogenic built-up areas and the high-density of water mongooses in natural, fragmented habitats illustrates an inflexibility to deal with human disturbance. Water mongoose species in the present study highlight the general adaptive nature of the species in an urban environment. Anthropogenic habitat transformation is a significant threat to the species, which is indicative of mesocarnivores that are reliant on green spaces in an urban matrix. However, these trends are not species-specific and can vary between locations (Kiawah bobcats vs general bobcat land-use) (Roberts 2007). The water mongoose is unlikely to become a true urban exploiter but instead, persist in the remaining natural fragments in or on the edge of the urban matrix landscape

Further research on the population genetics and health of urban mongoose species is necessary for understanding their population dynamics and their ability to persist in the urban matrix landscape. Lastly, there is a need to increase urban ecological research output on mesocarnivores in South Africa and Africa. Current literature is species and geographically limited, leaving most of the world's urban (developing countries) mesocarnivores relatively unstudied.

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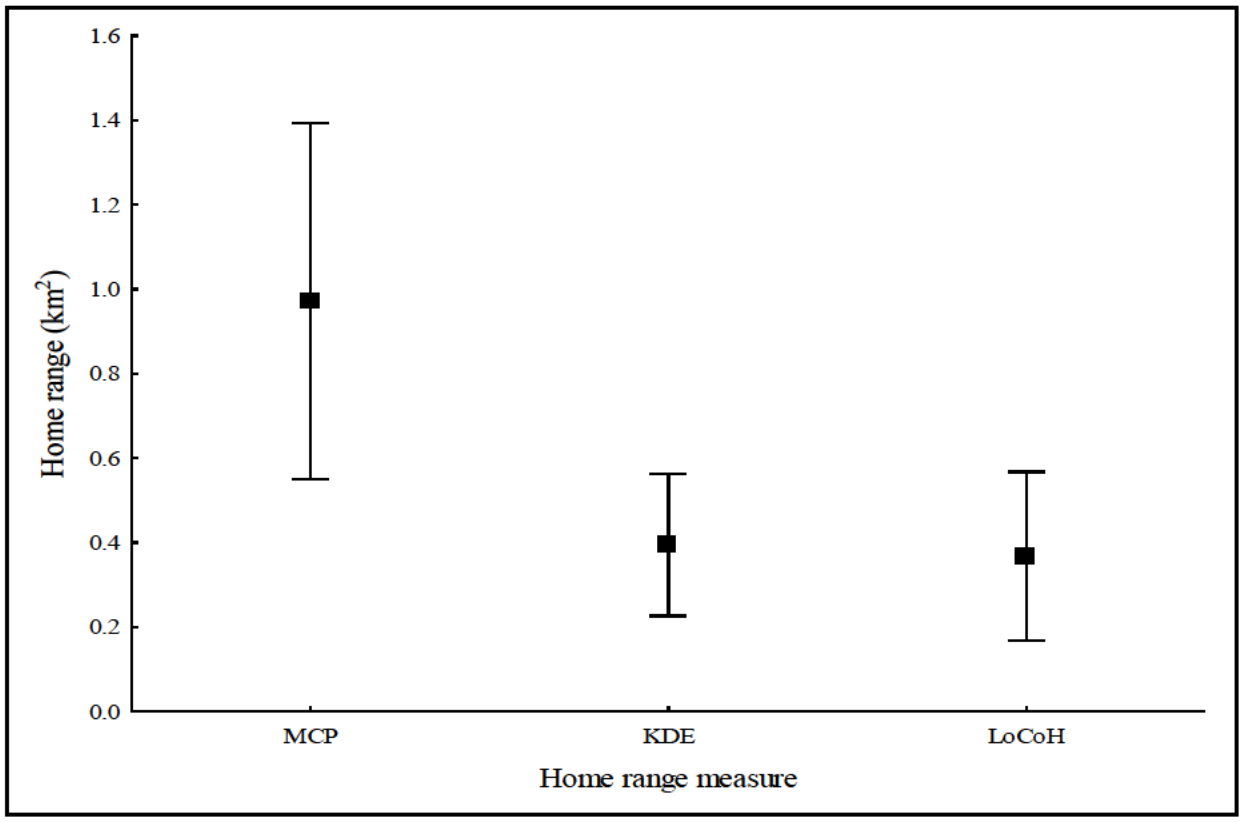
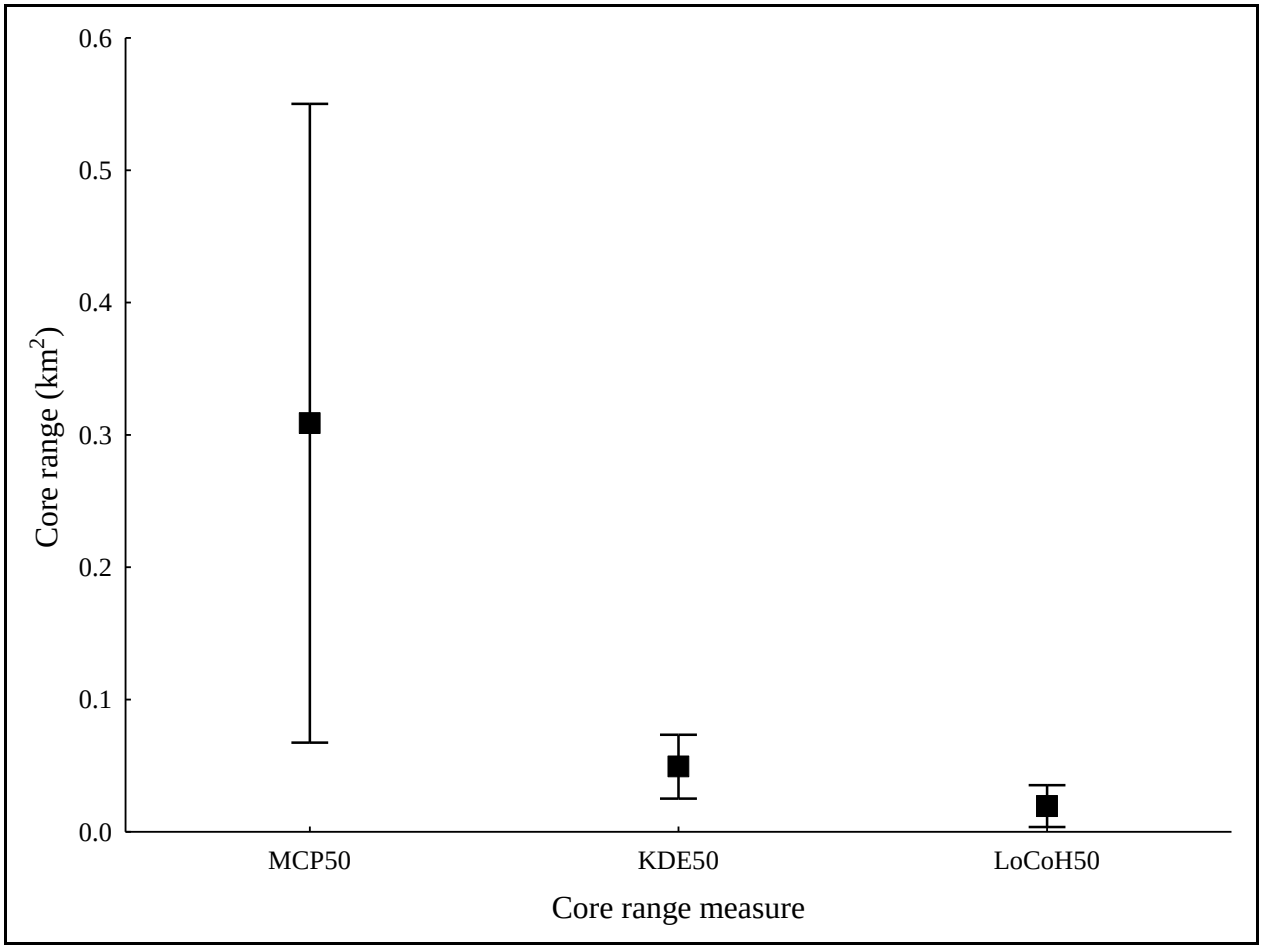
## 4.8 Supplementary information




**Supplementary information Figure S4. 1.** Comparison of three different home range estimation methods (95% MCP, KDE and LoCoH) of overall home ranges for water mongoose in the present

study.




**Supplementary information Figure S4. 2.** Comparison of three different core range estimation methods (50% MCP, KDE and LoCoH) of overall core range for water mongoose in the present study.

**Supplementary information Table S4. 1.** Habitat use of water mongoose from expected proportion of habitat availability and proportion of habitat utilised, using Bonferroni confidence intervals within 100% Maximum Convex Polygon (MCP) home range in Kloof, eThekwini, Durban, South Africa in the present study.

| | | | Habitat | | | |
|---|---|---|---|---|---|---|
| **Mongoose ID** | **Measure** | **Grassland** | **Forest and bushland** | **Water** | **Urban** | **Plantation** |
| W1 | Availability | NIL | **0.335** | **0.101** | **0.000** | **0.012** |
| | Observed use | NIL | 0.613 | 0.172 | 0.136 | 0.079 |
| | Bonferroni | NIL | 0.57 < P < 0.656 | 0.139 < P < 0.205 | 0.105 < P < 0.166 | 0.055 < P < 0.103 |
| W2 | Availability | **0.005** | **0.559** | 0.060 | **0.357** | 0.019 |
| | Observed use | 0.000 | 0.782 | 0.087 | 0.102 | 0.029 |
| | Bonferroni | -0.001 < P < 0.001 | 0.725 < P < 0.838 | 0.049 < P < 0.126 | 0.061 < P < 0.143 | 0.006 < P < 0.052 |
| W3 | Availability | 0.006 | **0.567** | **0.089** | **0.338** | NIL |
| | Observed use | 0.003 | 0.668 | 0.163 | 0.166 | NIL |
| | Bonferroni | -0.003 < P < 0.009 | 0.616 < P < 0.719 | 0.123 < P < 0.203 | 0.126 < P < 0.207 | NIL |
| W4 | Availability | NIL | **0.316** | **0.033** | **0.612** | **0.039** |
| | Observed use | NIL | 0.604 | 0.139 | 0.243 | 0.014 |
| | Bonferroni | NIL | 0.524 < P < 0.684 | 0.082 < P < 0.195 | 0.173 < P < 0.313 | -0.005 < P < 0.033 |
| W5 | Availability | **0.009** | **0.567** | **0.108** | **0.316** | NIL |
| | Observed use | 0.000 | 0.720 | 0.054 | 0.226 | NIL |
| | Bonferroni | -0.002 < P < 0.002 | 0.629 < P < 0.812 | 0.008 < P < 0.1 | 0.141 < P < 0.311 | NIL |
| W6 | Availability | **0.012** | 0.490 | **0.104** | **0.262** | 0.133 |
| | Observed use | 0.000 | 0.445 | 0.293 | 0.110 | 0.152 |
| | Bonferroni | -0.001 < P < 0.001 | 0.369 < P < 0.521 | 0.223 < P < 0.362 | 0.062 < P < 0.158 | 0.097 < P < 0.207 |
| W7 | Availability | NIL | **0.500** | **0.104** | **0.396** | NIL |
| | Observed use | NIL | 0.900 | 0.040 | 0.060 | NIL |
| | Bonferroni | NIL | 0.817 < P < 0.983 | -0.014 < P < 0.094 | -0.006 < P < 0 126 | NIL |
| W8 | Availability | NIL | **0.444** | 0.045 | **0.511** | NIL |
| | Observed use | NIL | 0.697 | 0.059 | 0.243 | NIL |
| | Bonferroni | NIL | 0.631 < P < 0.764 | 0.025 < P < 0.094 | 0.181 < P < 0.305 | NIL |
| W9 | Availability | **0.007** | **0.260** | **0.034** | **0.649** | 0.050 |
| | Observed use | 0.000 | 0.583 | 0.078 | 0.307 | 0.032 |
| | Bonferroni | -0.001 < P < 0.001 | 0.531 < P < 0.635 | 0.05 < P < 0.107 | 0.259 < P < 0.356 | 0.013 < P < 0.05 |
| W10 | Availability | NIL | NIL | NIL | NIL | NIL |
| | Observed use | NIL | NIL | NIL | NIL | NIL |
| | Bonferroni | NIL | NIL | NIL | NIL | NIL |
| W11 | Availability | NIL | NIL | NIL | NIL | NIL |
| | Observed use | NIL | NIL | NIL | NIL | NIL |
| | Bonferroni | NIL | NIL | NIL | NIL | NIL |
| W12 | Availability | **0.063** | 0.458 | **0.071** | **0.408** | NIL |
| | Observed use | 0.000 | 0.487 | 0.355 | 0.158 | NIL |
| | Bonferroni | -0.002 < P < 0.003 | 0.374 < P < 0.599 | 0.248 < P < 0.463 | 0.076 < P < 0.24 | NIL |
| W13 | Availability | 0.002 | 0.733 | **0.061** | **0.131** | 0.073 |
| | Observed use | 0.000 | 0.908 | 0.011 | 0.046 | 0.034 |
| | Bonferroni | -0.002 < P < 0.002 | 0.847 < P < 0.969 | -0.011 < P < 0.034 | 0.002 < P < 0.09 | -0.004 < P < 0.073 |
| W14 | Availability | NIL | NIL | NIL | NIL | NIL |
| | Observed use | NIL | NIL | NIL | NIL | NIL |
| | Bonferroni | NIL | NIL | NIL | NIL | NIL |

# CHAPTER 5

## Community perceptions of mammalian mesocarnivores across a land-use gradient in KwaZulu-Natal, South Africa

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**Running header:** Citizens' perceptions of mammalian mesocarnivores

## 5.1 Abstract

Globally anthropogenic land conversion has resulted in reduced natural land available for free-roaming wildlife and a shift in species composition. Certain mesocarnivores exhibit an ability to adapt and expand distribution ranges under differing anthropogenic pressures. Our study addressed the socio-ecological attitudes and general perspectives towards mammalian mesocarnivores across a land-use gradient (rural-urban) from the uMgungundlovu to eThekwini Municipalities of KwaZulu-Natal, South Africa. We determined threats, conflicts and general attitudes towards mammalian mesocarnivores using an online questionnaire survey. The public held a range of opinions on mesocarnivores. Participants from both land-use types generally regarded mesocarnivores as non-threatening and vital for the environment. However, not all mesocarnivores were regarded with equal benevolence. Black-back jackals *Canis mesomelas* were consistently singled out as a problematic species by rural inhabitants and this intensified reactions when a confrontation occurred. Smaller and more behaviourally flexible mesocarnivores (mongoose species and genets *Genetta* spp.) were the most observed species in both land-use types demonstrates the species adaptive abilities. Vehicle collision was the leading cause of mesocarnivore fatalities. Overall perceived impacts and threats of mesocarnivores differed across the land-use gradients. Our study illustrates the importance of citizen science and community engagement across a land-use gradient in wildlife conservation efforts.

**Keywords:** Citizen Science, Human–wildlife interactions, land-use gradient, mammal, mesocarnivore, Online wildlife survey

## 5.2 Introduction

Carnivore conservation remains a challenge for those tasked with their management (Ripple *et al.*, 2014). The ecological requirements of carnivores and the pervasiveness of human–wildlife interactions (HWI) are complications conservation practitioners face (Gittleman *et al.*, 2001). In an ideal scenario, carnivore conservation would aim to conserve a sufficiently viable population within its species-specific ecological requirements (Woodroffe & Ginsberg, 1998). However, with ever-increasing global anthropogenic land-use change, this ideological goal is becoming unattainable. Approximately 7.5% of the world's terrestrial surface falls under formal protected and continuous landscape (Saura *et al.*, 2018). The processes of habitat loss by anthropogenic land-use change (urbanisation and agriculture) are suggested to be some of the greatest threats to global biodiversity (Sala *et al.*, 2000; Newbold, 2018; Powers & Jetz, 2019). Land-use change radically transforms and fragments natural landscapes for human exploits (Haddad *et al.*, 2015). With the human ecological footprint compounding, distribution ranges between wildlife and humans inevitably overlap, thus mounting HWI (Madden, 2004).

The human-dominated land transformation has altered natural habitats (fragmentation, degradation and patch sizes) negatively, which has affected habitat use by wildlife. The distribution range of many mammalian mesocarnivores in South Africa has received limited attention (Skinner & Chimimba, 2005). However, there is a growing body of evidence that several species of generalist, medium–sized carnivores are increasing their range and distribution in the absence of large predators under varying land-use type (Ramesh, Kalle & Downs, 2015; Humphries *et al.*, 2016; Ramesh, Kalle & Downs, 2016; Widdows & Downs, 2016; Loock *et al.*, 2018; Ponsonby & Schwaibold, 2018; Cronk & Pillay, 2019; Cronk & Pillay, 2020; Leighton *et al.*, 2020). Mesocarnivores are often classified as generalist and opportunistic feeders with the ability to occupy, and in some instances, colonise a variety of habitats (Prugh *et al.*, 2009). Although mesocarnivores are generally suggested to be highly

adaptable to dynamic environmental change, they are equally vulnerable to HWIs because of the relationship with resource availability (Ramesh, Kalle & Downs, 2016).

Mesocarnivores are attracted to anthropogenic resources (livestock, shelter, refuses, breeding sites) in human-modified landscapes, and this is often compounded by the loss of their native habitat and continued depletion of wild prey (Gehrt, Riley & Cypher, 2010). Within South African farming communities livestock husbandry is a vital socio-economic process (Kerley, Wilson & Balfour, 2018). Livestock depredation, particularly by caracal (*Caracal caracal*) and black-backed jackal (*Canis mesomelas*), often causes substantial economic losses to farmers, frequently resulting in HWI (Humphries, Hill & Downs, 2015; Kerley, Wilson & Balfour, 2018). These factors generally result in negative attitudes of local farming communities towards specific mesocarnivores species (Kerley, Wilson & Balfour, 2018). The tolerance of commercial farmers towards these impacts are broadly shaped by socio-economic factors (wealth, levels of education, wildlife derived impact costs, personal values and monetary benefits vs relative losses) (Zimmermann, Walpole & Leader-Williams, 2005). A farmer from Humphries *et al.* (2015) study lost a total of 260 sheep and 151 lambs over three years because of depredation. This subsequently led to retaliation against a suspected species, the black-backed jackal, which the farmer shot 144 individuals over that same period. It is apparent the farmers often resort to uncontrolled and unsustainable management methods (poisoning and shooting) in an attempt to eradicate problematic species which often lacks species specificity (Kerley, Wilson & Balfour, 2018). The HWI relationship between farmers or perceived vermin further perpetuates negative attitudes that ultimately hinders mesocarnivore conservation (Kruuk, 2002; Humphries, Hill & Downs, 2015; Plessis, Avenant & Waal, 2015).

Interactions between these generalist mammalian carnivores and humans are becoming frequent (Bateman & Fleming, 2012; Šálek, Drahníková & Tkadlec, 2015). A concerted effort is required to understand the relationship that exists between mesocarnivores and humans. Humans are a central part of modified landscapes; therefore, it is crucial to incorporate their perspectives in

conservation measures (McIntyre, Knowles-Yanez & Hope, 2008; Dickman, 2010). The use of survey questionnaires has been a growing technique in applied ecology (Grey-Ross, Downs & Kirkman, 2010; Widdows, Ramesh & Downs, 2015). Community-based wildlife surveys allow for a rapid accumulation of vital information on the presence and abundance of commonly identified species (FitzGibbon & Jones, 2006), and the human perspective toward species. Historical trends in species diversity and richness can be derived from residents that have lived in areas for extended periods (White *et al.*, 2005). Community-based questionnaires provide essential insights into prospective management programmes and their likelihood of acceptance and subsequent success within communities (White *et al.*, 2005; FitzGibbon & Jones, 2006; Dickman, 2010). Community engagement through questionnaires prompts the expression of public opinions while simultaneously enhancing social awareness and wildlife education (Dickman, 2010). Additionally, systematic community-based surveys provide qualitative historical and real-time information of anthropogenic impacts that can be incorporated in the establishment of most promising management practices to effectively manage wildlife (Frank, Johansson & Flykt, 2015; Widdows, Ramesh & Downs, 2015). The above factors highlight the usefulness of community-based questionnaires in minimising negative interactions and bolstering overall wildlife conservation efforts.

The landscape changes in KwaZulu-Natal Province, South Africa, have exposed mesocarnivores to extensive habitat modification, prompting a dependency for overlapping resources (space, resources and livestock). The extent to which habitat modification has occurred differs along a land-use gradient and therefore the subsequent levels human-wildlife interactions/impacts. The successful preservation of wildlife needs to recognise, consider, and translate human perceptions into wildlife management under differing landscape pressures (Natural-Urban). To understand the socio-ecological factors and attitudes towards mammalian mesocarnivores, we conducted a questionnaire survey with the public along a land-use gradient from the Midlands to the eThekwini Metropol in KwaZulu-Natal. We aimed to determine the factors that influenced perspectives towards

mesocarnivores and the threats mesocarnivores face. We assessed attitudes, threats and interactions of the general public. We were particularly interested in how opinions vary and how mesocarnivores affect the two groups (Rural vs Urban). Additionally, does the socio-economic background influence knowledge on species and positive/negative attitude? We hypothesised that there would be a range of attitudes towards mesocarnivores. We predicted that opinions would differ between farm owners and those residing in urban areas. We generally expected positive opinions from people in urban areas because they are less affected by mesocarnivores than farmland residents. Conversely, farmland residents would likely have a more mutualistic attitude towards mesocarnivores. We expected rural residing respondents would have more interactions with a higher diversity of mesocarnivores. Whereas smaller sized, more generalist mesocarnivores species would likely be frequently sighted in urban areas. Additionally, we predicted that the anthropogenic threats of vehicle collisions caused mortality for mesocarnivores would be higher in urban areas compared with rural areas. Overall, we hoped to identify common areas of conflict along the land-use gradient to supplement future wildlife management strategies in KwaZulu-Natal.

## 5.3 Methods

*Study sites*

KwaZulu-Natal has the second largest human population out of South Africa's nine provinces (Statistics South Africa, 2018). The municipalities of focus for the questionnaire survey were the metropole of eThekwini and district of uMgungundlovu (Fig. 5.1). The eThekwini Metropolitan is on the eastern seaboard of South Africa and contains the largest city in the province, namely Durban (Statistics South Africa, 2018). The uMgungundlovu district forms part of the KwaZulu-Natal Midlands and includes the second-largest city in the province, which is Pietermaritzburg (Statistics South Africa, 2018). The physical geography of these municipalities is a mixture of heavily urbanised industries to natural farmland mosaic on the edge of the KwaZulu-Natal Drakensberg (Statistics South

Africa, 2018). Both municipalities house a diverse range of social classes of different cultural and economic standing. The standard of living in these areas ranges from formal to informal and traditional dwellings in both rural and dense urban living spaces (Statistics South Africa, 2018). Rural and urban areas were classified based on census information from Statistics South Africa (Statistics South Africa, 2018)

The study region is situated within a global biodiversity hotspot, the Maputaland-Pondoland Albany (MPA) (Steenkamp *et al.*, 2004). The biodiversity-rich natural vegetation consists of mesic grasslands, savannas, forests and wetlands with high levels of endemism (Mucina & Rutherford, 2006). Natural greenbelts and conservancies are interspersed within the diverse urban matrix (Thabethe & Downs, 2018). The municipalities have rapidly developed in the past decades (McPherson, Brown & Downs, 2016; Singh & Downs, 2016). In particular, eThekwini's natural vegetation has undergone considerable transformation, with two-thirds of its original distribution having experienced anthropogenic transformation (EPCPD, 2015). However, Ethekwini Municipality has the Durban Metropolitan Open Space System (D'MOSS) which is a system of green corridors that link essential biodiversity conservation sites in eThekwini to facilitate passage of fauna and flora (Roberts, 1994). The KwaZulu-Natal western landscape experiences Drakensberg mountain climes and subtropical climes of the Indian ocean in the east (Jewitt *et al.*, 2015). A diverse range of mammals occurs along this land-use gradient from natural habitat to the densely populated urban matrix within the municipalities (Rowe-Rowe, 1992; Rowe-Rowe, 1994).

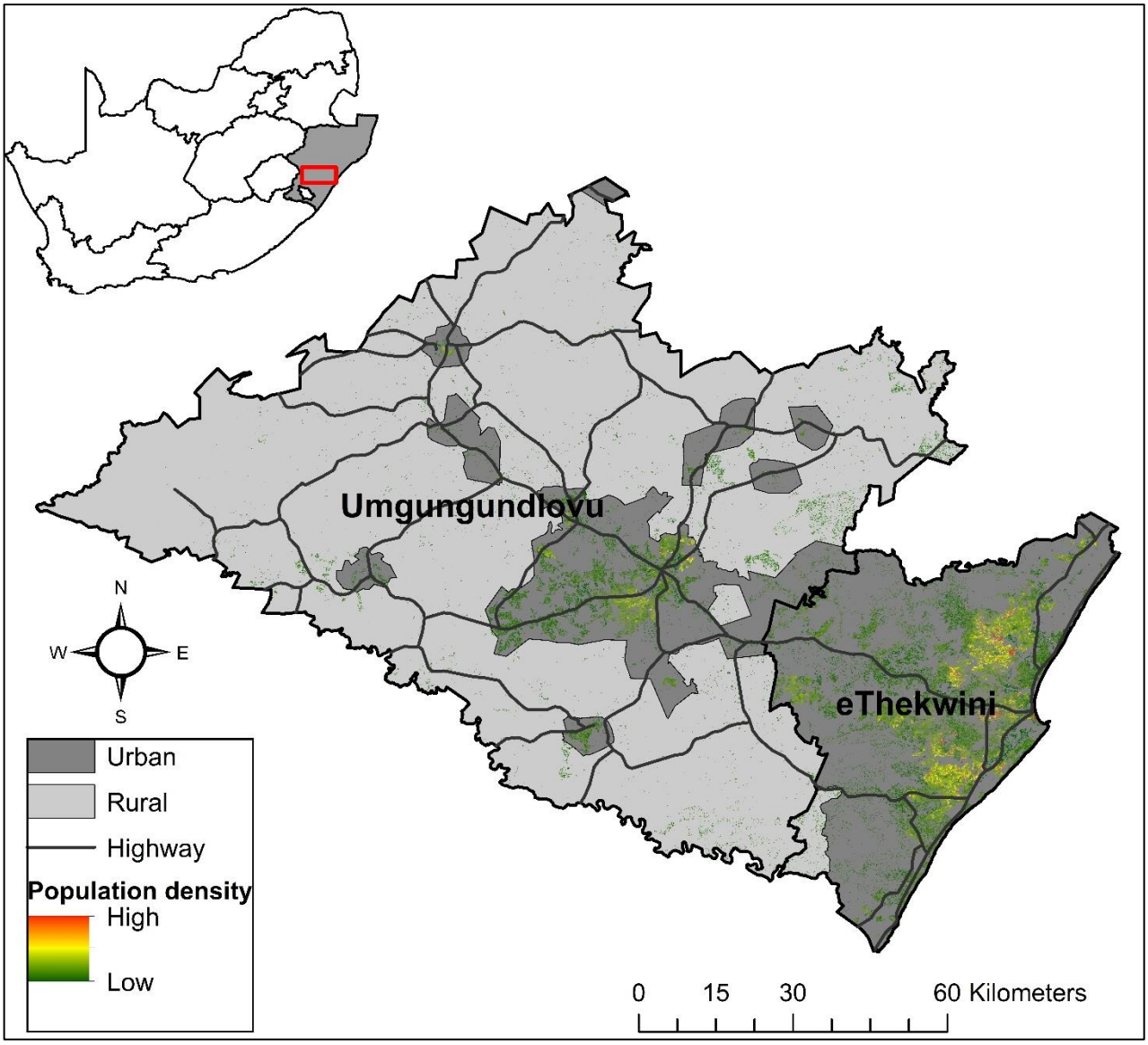


**Figure 5. 1.** The land-use transition and population density between the two districts (uMgungundlovu and eThekwini) in which the present survey was conducted in KwaZulu-Natal Province, South Africa.

*Survey and participant recruitment*

From January 2019 – January 2020 a questionnaire survey, generated on Google forms, was widely distributed on popular media platforms (Email, Facebook, conservancy newsletters and newspapers) and at community engagement meetings throughout KwaZulu-Natal. We also emailed the survey to various farming communities and conservancies in KwaZulu-Natal. We made the questionnaire freely sharable to reach a diverse range of participants from all walks of life. Attached to the survey was a

letter in English and isiZulu explaining the purpose of the study and motivating to forward it to others. We regularly sent follow up emails, Facebook posts, and had newspaper inserts to increase response rate and involvement from the community as suggested in other studies (Sheehan, 2001). Electronic questionnaire surveys have numerous benefits over interviews and other qualitative methods. The benefits include cost efficiency, speed of data collection, and larger data sets in a short duration without interviewer bias (Sheehan, 2001). However, electronic surveys are limited by internet access and computer literacy.

Our survey consisted of 37 questions divided into three sections: Section A: Participant details; Section B: Perceptions of medium–sized mammalian carnivores and Section C: Additional comments. Sociodynamic questions were initially requested from participants (gender, age, occupation, etc.). Participants were then asked questions on the abundance, richness and perspectives of mesocarnivores species found in their area of living. The survey was designed to enable self-completion (Supplementary information Appendix 5.A). The format style allowed participants to answer questions in their own time and space. The questionnaire comprised of three question styles, including dichotomous questions (yes/no), multiple-choice questions and open-ended questions. The open-ended questions were asked to gain personal attitudes and detailed comments that are not possible through dichotomous and multiple-choice questions. Ethical clearance in line with the ethical standards of the national and institutional committees on human experimentation of the Helsinki Declaration of 1975 (revised in 2008) was obtained from University of KwaZulu-Natal (Protocol number HSS/0909/018 D).

*Statistical analyses*

The exploratory nature of the study necessitated the use of only descriptive and non-parametric statistics. Pearson's chi-square tests ($\chi^2$) were used to determine differences in respondents' perceptions of mesocarnivores based questions in relations to land-use type (rural vs urban). Moreover, Chi-square tests were performed to determine difference within land-use types (e.g., major cause of

mesocarnivore fatality in rural or urban). All statistical analyses were performed in SPSS (IMB SPSS version 27).

## 5.4 Results

*Demographics of participants*

A total of 203 people participated in the online questionnaire. The primary land-use type which the respondents reside in varied in the two districts (uMgungundlovu and eThekwini). The majority of participants 159 (78%) were from urban/peri-urban areas, and 44 (22%) from farmland/rural areas. The age profile of respondents ranged, with 22 (11%) 18-24 years old, 44 (22%) 25-34 years old, 49 (24%) 35-49 years old, 73 (36%) 50-70 years old and only 15 (7%) older than 70 years. A total of 34 (17%) respondents were unemployed or had part-time employment (Table 5.1). The remaining 169 (83%) respondents were either self-employed, employed, or studying. The majority of respondents indicated that they were not actively involved in a wildlife conservancy (67%). However, participants felt that being a member of one would have a positive impact on wildlife conservation ($\chi^2 = 184.95$ df $= 2$, $p < 0.05$). The positive attitude towards a personal active engagement in a conservancy was consistent across the lands-use types (Urban: $\chi^2 = 138.83$ df $= 2$, $p < 0.05$, Rural: $\chi^2 = 46.68$ df $= 2$, $p < 0.05$; Figure 5.2a).

**Table 5.1.** Demography of 203 respondents on the perspectives of mammalian mesocarnivores across a land-use gradient (rural–urban) in two districts (uMgungundlovu and eThekwini) in KwaZulu-Natal, South Africa.

| | Land-use type (n) | | Total Percentage |
|---|---|---|---|
| | **Rural** | **Urban** | |
| **Gender** | | | |
| Female | 19 | 82 | 50 |
| Male | 25 | 76 | 50 |
| Prefer not to say | 0 | 1 | 0 |
| **Age Profile (years)** | | | |
| 18-24 | 6 | 16 | 11 |
| 25-34 | 4 | 40 | 22 |
| 35-49 | 6 | 43 | 24 |
| 50-70 | 23 | 50 | 36 |
| 70+ | 5 | 10 | 7 |
| **Race** | | | |
| Asian | 0 | 3 | 1 |
| Black African | 6 | 14 | 10 |
| Indian | 1 | 7 | 3 |
| Other | 0 | 3 | 2 |
| White | 37 | 132 | 83 |
| **Land-use type** | | | |
| Farmland & rural | 44 | 0 | 22 |
| Urban and peri-urban | 0 | 159 | 78 |
| **Employment** | | | |
| Full time | 19 | 75 | 46 |
| Unemployed | 1 | 16 | 8 |
| Part-time | 6 | 11 | 8 |
| Self-employed | 13 | 30 | 21 |
| Student | 5 | 27 | 16 |
| **Highest level of education** | | | |
| Matric/Grade 12 | 8 | 27 | 17 |
| Post School diploma | 7 | 28 | 17 |
| Tertiary degree | 29 | 28 | 66 |

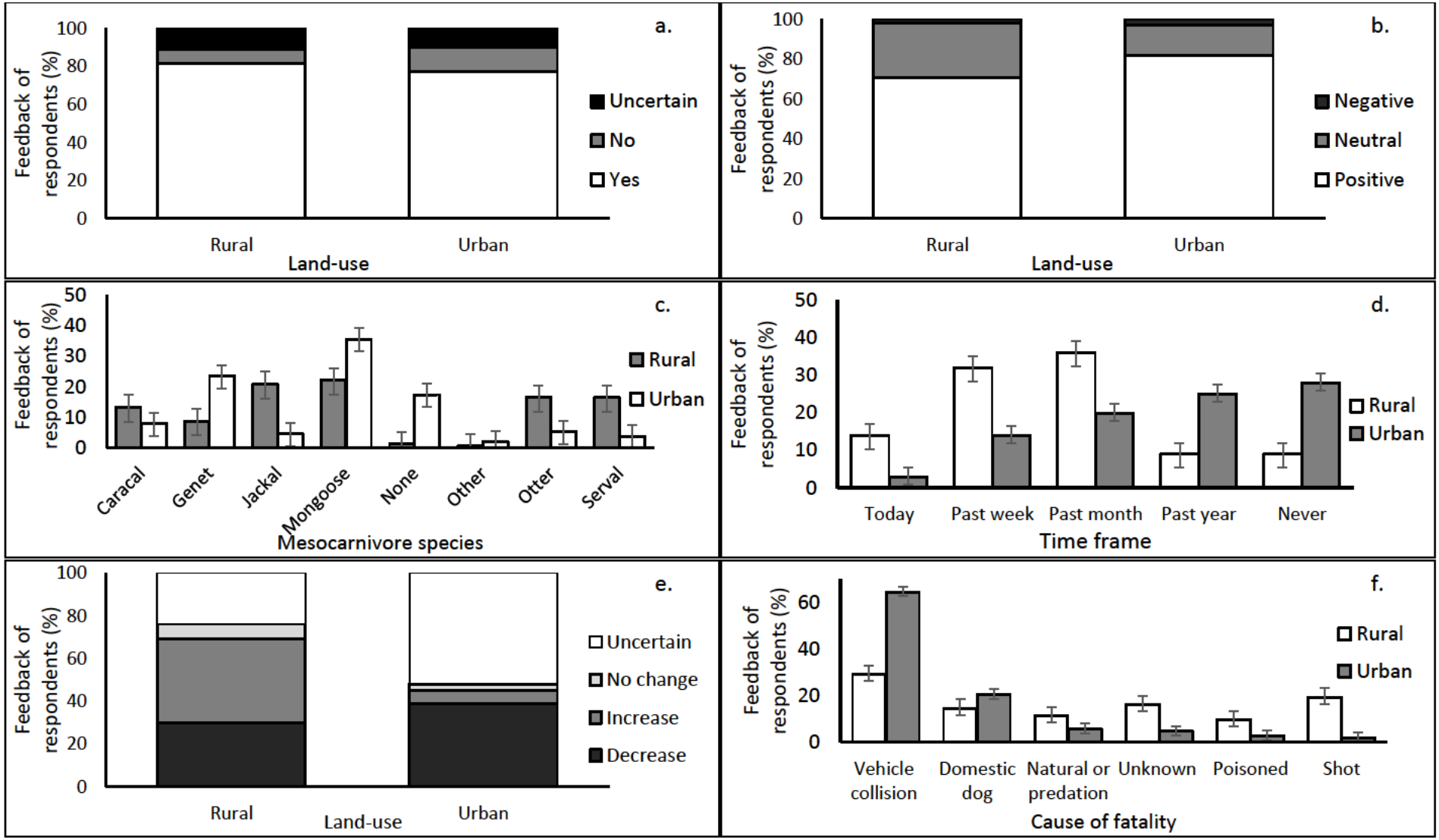


**Figure 5.2.** a. Perceptions of active engagement in a wildlife conservancy leading to the improvement of wildlife conservation. b. Perceptions of mammalian mesocarnivores. c. Mammalian mesocarnivores sighted by participants of the survey. d. Time frame from when last participants saw evidence of a mammalian mesocarnivore in their routine activities. e. The perceived changes in mesocarnivore populations over ten years. f. Feedback of respondents of the possible cause of fatalities in mammalian mesocarnivores with 95% confidence intervals in each land-use type (rural–urban) by respondents in this study.

*Perceptions of mammalian mesocarnivores*

Many (67%) participants understood what a mammalian mesocarnivore was. There was no difference in the level of understanding of what a mammalian mesocarnivore is between the rural and urban categories ($\chi^2 = 2.68$, df = 3, $p > 0.05$). Respondents perceived mesocarnivores differently between the two land-use types ($\chi^2 = 12.27$, df = 2, $p < 0.05$). The majority of responses from both land-use types (Rural: 28 (64%), Urban: 134 people (84%)) shared a positive stance towards mesocarnivores followed by neutral and negative ones, respectively (Rural: $\chi 2 = 20.36$, df = 2, $p < 0.005$, Urban: $\chi 2 = 189.85$, df = 2, $p < 0.005$; Figure 5.2b). Five participants (Rural: 3, Urban:2) had negative perceptions of mesocarnivores.

The most commonly sighted mesocarnivores across both land-use types were mongooses with variations in the frequency of sightings of other species (Figure 5.2c). There was a difference in the species composition sighted between Rural and Urban category ($\chi^2 = 100.53$, df = 7, $p < 0.005$). Larger mesocarnivores (caracal, jackal, otter: *Aonyx capensis*, and serval: *Leptailurus serval*) were frequently sighted in the rural land-use type (Figure 5.2c). Smaller mesocarnivores (cape genet and mongooses: *Galerella sanguinea*, *Atilax paludinosus*, *Ichneumia albicauda*) were regularly sighted in both rural and urban landscapes (Figure 5.2c). Six respondents indicated sighting of species that were not listed (Rural = 1, Urban = 5). African striped weasel *Poecilogale albinucha*, and free-roaming domestic cats *Felis catus* and dogs *Canis lupus familiaris* in urban and African civet *Civettictis civetta* in rural areas. Caracals were sighted numerous times by respondents in urban areas from which caracal were historically absent.

The time category since last sighting a mesocarnivore varied between the land-use types ($\chi^2 = 32.01$, df = 4, $p < 0.005$). Respondents from rural areas had more frequent encounters with mesocarnivores than urban-residing respondents, which indicated lengthier lapse between sightings Figure 5.2.d). The main behavioural activity of mesocarnivores reported during

sightings was foraging and hunting (Rural 54%, Urban 36%), followed by moving and crossing roads for both rural and urban areas (Table 5.2). Respondents from rural areas observed mesocarnivores regularly throughout the year ($\chi^2 = 2.52$, df = 3, $p > 0.05$) whereas participants from urban areas indicated that sightings occurred during warmer months (summer (44%) and spring (31%) ($\chi^2 = 23.00$, df = 3, $p < 0.005$), and not during cooler months of the year.

**Table 5.2.** The observations and frequencies of activities of mesocarnivores by respondents in two differing land-use types in the uMgungundlovu and eThekwini districts in KwaZulu-Natal, South Africa. Results questionnaire survey reported between 2018–2019.

| | N | | Frequency (%) | |
|---|---|---|---|---|
| **Observations** | **Rural** | **Urban** | **Rural** | **Urban** |
| Calling | 3 | 2 | 5.9 | 1.7 |
| Crossing road | 9 | 20 | 17.6 | 16.8 |
| Foraging/hunting | 28 | 43 | 54.9 | 36.1 |
| Moving | 10 | 41 | 19.6 | 34.5 |
| Scat/spoor | 1 | 5 | 2.0 | 4.2 |
| Dead | 0 | 4 | 0.0 | 3.4 |
| Sitting | 0 | 4 | 0.0 | 3.4 |

*Impacts and threats*

There were differing views between rural and urban respondents when asked whether mesocarnivore population numbers have shifted in the last ten years ($\chi^2 = 34.53$, df = 3, $p < 0.005$). Interestingly there were high opposing views in the rural land-use type. The majority of respondents (39%) indicated there had been an increase, followed by a decrease (30%) in mesocarnivore populations. More than half (52%) of the responses from urban areas indicted that they were overall uncertainty about historical population numbers of mesocarnivores (Figure 5.2e). Black-backed jackal was purported to be the species that had increased in population size in rural areas over the past ten years.

There was, however, a contrast in the perceptions of mesocarnivores as carriers of zoonotic disease between rural and urban areas ($\chi 2 = 18.53$, df = 2, $p < 0.005$). A higher percentage of participants from rural areas (64%) indicated that mesocarnivores are active carriers of zoonotic diseases compared with the urban areas (29%). Generally, there was considerable (40%) uncertainty over mesocarnivores as carriers of zoonotic disease within the urban sector. A total of 77 respondents (rural: 30, urban: 47) indicated concerns related to Rabies lyssavirus, with minimal concerns regarding tick-borne diseases (rural: 1, urban: 3), sarcoptic mange (*Sarcoptes scabiei*) (rural: 2, urban: 2), and *Echinococcus* (rural: 1, urban: 1). Overall participants from both land-use types viewed living in an area inhabited by mesocarnivores as non-threatening (rural: 84%, urban 92%) ($\chi 2 = 2.33$, df = 2, $p > 0.05$). Participants that did feel threatened (rural: 16%, urban 8%) by mesocarnivores residing near-human (homesteads) indicated fears of being a victim of an attack or becoming infected with rabies.

Respondents perceived the effects of mesocarnivores on the wildlife differently between the two land-use types ($\chi 2 = 40.10$, df = 3, $p < 0.005$). Participants from urban areas indicated that the presence of mesocarnivores had a positive effect on the surrounding wildlife (53%) as opposed to in rural (27%). Notably, the presence of mesocarnivores was suggested to play an integral role in maintaining ecosystem balance with specific reference to the control of pest species. There was a high degree of neutrality shared between the two land-use types (rural: 43%, urban 46%). In contrast, 30% of rural participants felt that mesocarnivores had a negative impact on wildlife. Black-backed jackal was singled out as the problematic species because of perceptions of uncontrolled population expansion and predation of livestock. Differences in occurrences of intentional supplementary feeding of mesocarnivores were not present in either land-use type ($\chi 2 = 2.30$, df = 3, $p > 0.5$). Only four respondents from urban areas indicated that they supplementary fed mesocarnivores with the majority of respondent

never feeding in either rural 41 (93%) or urban 150 (94%). Mongooses and genets were the only taxa that were being intentionally fed by participants. One dairy farmer indicated that jackals' diets are supplemented by calving dairy cows throughout the year.

Mesocarnivore fatalities appeared widespread in both land-use types. However, no difference in the occurrence of mesocarnivore fatalities ($\chi2 = 5.59$, df = 2, $p > 0.5$) was detected. Participants from rural areas (70%) frequently indicated having seen a mesocarnivore fatality as compared with urban areas (51%). Vehicle collision was the principal cause of fatality for mesocarnivores in both rural and urban areas (Figure 5.2f). Eradicative control methods were commonly used in rural areas. These were either one of two methods: poisoning (10%) or shooting (16%) (Figure 5.4). Domestic pets were responsible for a fifth of fatalities in urban areas (Figure 5.2f).

Overall, significantly more participants from both land-use types (Rural: 89%, Urban: 87%) indicated that further mesocarnivore research is required ($\chi2 = 4.101$, df = 2, $p > 0.5$) (Rural: $\chi2=60.86$, df=2, $p < 0.005$, Urban: $\chi2 = 86.10$, df = 1, $p < 0.005$). A single participant from an urban lifestyle indicated no further research is required on mesocarnivores, and the participant indicated a negative attitude towards them.

## 5.5 Discussion

The incorporation and participation of communities is an essential step for successful future conservation strategies to protect mesocarnivores and the surrounding landscapes ultimately. The feedback received from the participants of two municipalities (uMgungundlovu and eThekwini) in KwaZulu-Natal, provides a preliminary perspective into similarities/differences in the attitudes, impacts and threats mammalian mesocarnivores face across a land-use gradient. The survey revealed that the majority of participants were not involved in any form of wildlife conservation. Although a wide range of perspectives, respondents commonly held

a positive interest in conserving local wildlife across the land-use types. The view that future involvement in a wildlife conservancy can result in the improvement in surround wildlife highlights the importance of community engagement as a practical step for stakeholders and environmental practitioners.

Smaller mesocarnivores of this study, namely mongooses and cape genet, were the most commonly occurring species across the land-use gradients. Although scientific literature is relatively sparse for these mesocarnivores in southern Africa, there is growing evidence that their general adaptability is either aiding their persistence or distribution range expansion (Widdows & Downs, 2016; Cronk & Pillay, 2019; Streicher, Ramesh & Downs, 2020). The widespread occurrence of smaller, behaviourally flexible mesocarnivore species indicates successful colonisation of certain urban areas in the absence of predator control. Urban ecological studies have revealed metropolitan carnivore species (red fox, raccoon, coyote: *Canis latrans* and European badger) have successfully established themselves globally, increasing in population densities in urban areas (Gehrt, Riley & Cypher, 2010; Bateman & Fleming, 2012; Šálek, Drahníková & Tkadlec, 2015). Rural and farmland regions are generally characterised by a more diverse array of carnivore species than urban cores areas (McKinney, 2002). However, Parsons *et al*. (2018) found that, when comparing between the land-use types, both species diversity and richness of carnivores in suburban and exurban areas was higher than or equal to, wild spaces. Suburban developed areas can benefit generalist carnivores through direct or indirect food resources and more diverse and varied landscapes compared with wild areas (Parsons *et al.,* 2018). However, the continued impact of anthropogenic land-use change, compounded by climate change, drives further imbalance within the system (Ripple *et al.*, 2017).

*Perceptions of mammalian carnivores*

In our study, participants from urban areas regarded mesocarnivores with a positive mindset compared with the widely differing views for rural inhabitants. It is commonly acknowledged that urban residents are more likely to support carnivore conservation than rural inhabitants (Morehouse *et al.*, 2020). Rural inhabitants live in closer proximity to carnivores and interact more frequently with them, often experiencing economic losses through livestock depredation. Therefore, mesocarnivores are less likely to face persecution in urban compared with rural areas. The perception within urban landscapes may have a positive influence on population expansion and distribution of urban adapted mesocarnivores in the future. A positive outlook towards urban carnivores is not unique to this study. Parallel perspectives towards carnivore species (coyotes, bobcat: *Lynx rufus*, red fox, European badger) are present globally within urban communities (Harrison, 1998; Sponarski *et al.*, 2016; Baker *et al.*, 2020). Generalist urban carnivores, such as red fox and raccoons, have expanded their distribution in urban areas of Britain, North America and considerable regions throughout Europe (Gehrt, Riley & Cypher, 2010). These two metropolitan generalist species have exploited the resource-abundant urban niche permitting them to expand their population densities and alter behavioural patterns (home range size, activity times, breeding patterns) compared with rural conspecifics. The urban environments in South Africa potentially provide an opportunity for behaviourally flexible mesocarnivore populations to establish.

There were an apparent disconnection and uncertainty in our study around the impacts, threats and population changes of mesocarnivores in urban areas compared with those from rural areas. This study shows that people from urban areas see mammalian mesocarnivores less frequently or never in their routine activities and have limited knowledge of changes in populations. Although there are various green spaces (Ezemvelo KZN Wildlife nature reserves,

conservancies and protected areas) throughout urban areas of this study, human–carnivore interactions appear to be scarce. In comparison, rural inhabitants seem to have more interaction and therefore, a deeper connection with native wildlife. Higher proportion of rural respondents perceived the impacts of mesocarnivores on surrounding wildlife negatively in this study. No further comment was available for participants to expand on their viewpoint. However, when asked if the presence of mesocarnivores benefits the community, there was a universal positive outlook towards them across the lands–use types. Surveyed residents expressed a deep-rooted appreciation for living in proximity to mesocarnivores, perceiving them as non-threatening. Mesocarnivores were viewed as species that prove vital for ecosystem services, namely pest control.

However, there was an evident distinction between the health threats that mesocarnivores can transmit between the land-use types. There appears to be a general uncertainty surrounding mammalian mesocarnivores as carriers of zoonotic diseases by urban participants. Rural inhabitants generally expressed concerns around the transmission of zoonosis, particularly rabies, with less concern expressed by urban residents. The general ignorance associated with zoonotic disease by urban respondents is of interest. An endemic strain of rabies does circulate within genets and mongooses (Nel, Roux & Atlas, 2009), whose populations have been suggested to be expanding in urban areas. However, the primary threat resides in canid transmission (black-backed jackal and domestic dogs) (Nel, Roux & Atlas, 2009). There have been increased efforts to raise public awareness of rabies in KwaZulu-Natal because the majority of the animal to human cases in South Africa have occurred in the province (Coetzee & Nel, 2007; Malerczyk *et al.*, 2010). However, the combination of urbanisation, increasing domestic dog populations and inadequate eradication measures continues to drive the persistence of rabies in KwaZulu-Natal, especially along the eastern seaboard (LeRoux *et al.*, 2018).

*Impacts and threats*

Mesocarnivores face numerous threats that are common in both land-use types. Vehicle collisions were the leading cause of fatality in this study. Road accidents are listed as a primary cause of anthropogenic caused mortality for mesocarnivores globally, regardless of land-use type (Bateman & Fleming, 2012). Haines *et al.* (2005) radio-tracking study of 80 ocelots *Leopardus pardalis* incurred a vehicle mortality rate of 45% of collared individuals. Similarly, roads have been the single cause of death for; European badgers in the UK (Clarke, White & Harris, 1998), bobcats in Illinois (Nielsen & Woolf, 2002), invasive raccoon dogs (*Nyctereutes procyonoides*) in Poland (Kowalczyk *et al.*, 2009), juvenile San Joaquin swift foxes (*Vulpes macrotis mutica*) in Kansas (Sovada *et al.*, 1998), and cape genets in KwaZulu-Natal (Widdows & Downs, 2018). Juveniles and males are more prone to this threat because of individual dispersal in their attempt to establish a defendable territory (Gehrt, Riley & Cypher, 2010). Road networks can act as barriers, preventing wildlife dispersal, particularly in heavily urbanised areas (Forman & Alexander, 1998). eThekwini, in particular, contains extensive road networks that bisect numerous D'MOSS areas. To cross into neighbouring green belt spaces, individuals must risk crossing roads and the chances of vehicles collisions increase. The impacts of roads are mitigated by the establishment of human–made culverts, overpasses and underpasses that allow wildlife to disperse unhindered by road networks (Clevenger, Chruszcz & Gunson, 2001).

Domestic pets were also responsible for higher rates of mortality in urban compared to rural areas. Widdows et al. (2017) indicated that domestics pets were the primary cause of fatalities in cape genets in Kloof (a residential area of eThekwini). Juvenile genets, in particular, were shown to be vulnerable to attack by domestic dogs (Widdows & Downs, 2018). Carnivore–domestic pet confrontation may potentially be providing a mode of disease

transmission in either direction (Mackenstedt, Jenkins & Romig, 2015). As residential areas sprawl, the consequences of these altercations may have dire repercussion in the future of wildlife and human health and the potential for spill-overs (Parrish *et al.*, 2008). A significant cause of mortality of mesocarnivores for this study in rural areas was by lethal measures (shooting and poisoning). Farmers frequently felt they had no choice but to resorted to several means of population control to attempt combatting pest animals and prevent economic strain through depredation of livestock. Despite the high presence of lethal control measures imposed on black-back jackals and other mesocarnivores populations, they continue to persist (Humphries, Hill & Downs, 2015). High population densities of black-backed jackals are a direct consequence of agriculture (concentration of food and poor disposal of carcasses) (Humphries, Hill & Downs, 2015). Lethal control may be an ineffective measure of population control for black-back jackal because removed individuals are simply replaced in high-density jackal populations in accordance with their social structure (Humphries, Hill & Downs, 2015; Minnie, Gaylard & Kerley, 2016). Deliberate control of problematic species may additionally result in "compensatory reproduction" which is present in several carnivore species including black-backed jackal (Canadian lynx: *Lynx canadensis*, Eurasian lynx: *Lynx lynx*, red fox and side-striped jackals: *Canis adustus*) (Parker *et al.*, 1983; Harris & Smith, 1987; Bingham & Purchase, 2002; Bagrade *et al.*, 2016; Minnie, Gaylard & Kerley, 2016). Large–scale compensatory reproduction manifests as increased fecundity, larger proportion of breeding females, increased reproductive lifespan, or a decrease in age at first reproduction of the targeted species (Kerley, Wilson & Balfour, 2018). Potentially ineffective and unsustainable population control measures coupled with concentrated food resources further exacerbates negative perceptions towards these species. Intense dislike towards black-back jackals is therefore expected to continue to persist within farming communities of KwaZulu-Natal.

*Limitations of the study*

Online questionnaire survey enables the rapid accumulation of information by a range of participants. However, online surveys are prone to several biases (sample validity, non-response and stakeholder bias) (Duda & Nobile, 2010). There were two major areas of concern of this study; firstly, the questionnaire was biased to those that have internet access. Although the study was distributed through various popular media platforms and potential respondents were encouraged at community engagement events, the result may not be accurately representative of the populations of two municipalities (uMgungundlovu and eThekwini) interests. Approximately 9% of the population of the KwaZulu-Natal Province have access to the internet within their dwelling (Statistics South Africa, 2016). Although the survey was limited by access to internet access, the two surveyed municipalities have the two highest levels of internet access for the province (Statistics South Africa, 2016). Secondly, the survey appears biased by the participants' race demography. Both regions are dominated by Black African populations (uMgungundlovu = 84.4% and eThekwini = 74.0%). Participants in this survey were mainly White, which highlighted a discrepancy in the actual reach of the survey. The participation of respondents of this survey highlights the bias encountered with online surveys because of South Africa's extreme poverty divide and Apartheid past.

## 5.6 Conclusions

Mammalian mesocarnivores within the two municipalities of uMgungundlovu and eThekwini are facing varying challenges based on their general adaptivity, behavioural plasticity and human persecution. Threats across the land-use types are largely anthropogenically driven (natural landscape conversion and lethal control). Mesocarnivore species appear to be more diverse and evenly distributed within rural/farmland systems. At the same time, more generalist and flexible mesocarnivores (mongooses and cape genets) are abundant in urban residential

systems. There exists an apparent appreciation for mesocarnivores and their roles within nature. However, not all species are viewed with an equal level of appreciation—black-backed jackal in particular face intense levels of animosity and persecution within farmland areas. The eagerness of participants to engage in wildlife initiatives indicates the need for cooperative efforts between various stakeholders. Wildlife education of residents across land-use types is imperative to conserving mesocarnivores in the future. However, one generic management strategy will not be suitable. Our study highlights that the different land-use types face and impose an array of threats on mesocarnivores.

Further research is required to reveal mesocarnivore ecological responses (behaviour, population size and density) under differing anthropogenic pressures throughout KwaZulu-Natal. A regional study on additional mesocarnivore species responses along a natural urban gradient will allow insight into the impact of human-altered landscapes on mesocarnivore ecology. In conclusion, future studies should incorporate in-person surveys to incorporate the perspectives of people, particularly in rural areas that do not have access to home-dwelling internet connectivity and are computer illiterate.

### 5.7 Acknowledgements

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**5.9 Supplementary information**

**Supplementary information Survey 5. 1.** Electronic questionnaire survey shared on by google documents on perceptions of medium–sized mammalian carnivores along a land-use gradient (rural-urban) in two municipalities (uMgungundlovu and eThekwini) in KwaZulu-Natal, South Africa

Date_____

What suburb and city do you live in?

_____

1: Section: Participants

1.1: Gender?

☐ Male ☐ Female ☐ Prefer not to say

1.2: What is your age group (in years)?

☐ 70+ ☐ 50-70 ☐ 35-49

☐ 25-34 ☐ 18-24

1.3: Race

☐ Black African ☐ Coloured ☐ White ☐ Asian

☐ Other

1.4: What Land-use area do you live in?

` ☐ Urban ☐ Peri-Urban ☐ Rural ☐ Farmland

☐ Other

1.5: Highest Level of Education

☐ None ☐ Primary ☐ Matric/Grade 12

☐ Post school certificate ☐ Tertiary (University Degree)

1.6: Occupation

☐ Full Time ☐ Part Time ☐ Self Employed

☐ Student ☐ Not employed

i. If "EMPLOYED" please indicate sector

_____

1.7: If you are a farmer, please specify below

☐ Crops ☐ Livestock/Dairy ☐ Poultry

☐ Dairy ☐ Subsistence ☐ Other

1.8: Are you a part of a wildlife conservancy?

☐ Yes ☐ No

i. If "Yes" please provide its name

_____

1.9: Regardless if you are not a part of a wildlife conservancy, do you feel being a part of one would promote a positive influence on your perspective of wildlife conservation?

☐ Yes ☐ No ☐ Uncertain

ii. Why do you feel this way?

_____

2. Section B: Perceptions of medium–sized mammalian carnivores

2.1: Do you know what a medium–sized mammalian carnivore is?

☐ Yes ☐ No ☐ Uncertain ☐ I don't know

2.2: Which of the following medium–sized mammalian carnivores have you seen before in your immediate area?

☐ Mongoose ☐ Caracal ☐ Jackal ☐ Otter

☐ Genet ☐ Serval ☐ Other ☐ None

2.3: If "Other" please specify species?

_____

2.4: **W**hen last did you see evidence of medium–sized mammalian carnivores in your day-to-day activities?

☐ Today ☐ This week ☐ This month ☐ This year ☐ Never

i. If so which species and how many?

_____

2.5: What activities did you see medium–sized mammalian carnivores doing (species-specific if possible)?

_____

2.6: In the last 10 years, do you feel medium–sized mammalian carnivores numbers have changed?

☐ Increase ☐ Decreased ☐ No Change ☐ Uncertain

2.7: If so, are certain medium–sized mammalian carnivores species' abundances increasing/decreasing more than others??

_____

2.8: Which season do you see the most medium–sized mammalian carnivores in your area?

☐ Spring ☐ Summer ☐ Autumn ☐ Winter ☐ Uncertain

2.9: What are your perceptions of medium–sized mammalian carnivores?

☐ Positive ☐ Neutral ☐ Negative

2.10: Have you in the past, or are you currently intentionally placing food out for medium–sized mammalian carnivores?

☐ Nightly ☐ Weekly ☐ Occasionally ☐ Never ☐ No Comment

i. If so, which species have you seen feeding?

_____

2.11: Have you ever seen a medium–sized mammalian carnivore fatality?

☐ Yes ☐ No ☐ Maybe

i. If yes state, the cause

☐ Domestic dog ☐ Vehicle collision ☐ Shot

☐ Poisoned ☐ Natural/predation ☐ Unknown

Do you think medium–sized mammalian carnivores carry diseases?

☐ Yes ☐ No ☐ Uncertain

ii. What diseases do you think medium–sized mammalian carnivores carry? If so, which species have you seen feeding?

_____

2.12: Do you feel that medium–sized mammalian carnivores are affecting other wildlife in your area?

☐ Yes ☐ No ☐ Uncertain

2.13: Do you feel at risk/threatened by living in an area that is inhabited by mammalian mesocarnivores?

☐ Yes ☐ No ☐ Uncertain

iii. What diseases do you think medium–sized mammalian carnivores carry? If so, which species have you seen feeding?

_____

2.14: Do you feel the community benefit by having mesocarnivores in the surrounding area?

☐ Yes ☐ No ☐ Uncertain

Why?

_____

Do you think more research should be performed on mesocarnivores?

☐ Yes ☐ No ☐ Uncertain

**Additional comments**

_____

# CHAPTER 6

## Diet of a generalist mammalian mesocarnivore species in an urban matrix

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**Running header:** Diet of urban water mongooses

## 6.1 Abstract

Anthropogenic habitat conversion through urban sprawl is forcing small carnivores to alter their behaviour and ecology. Thorough knowledge of their feeding ecology is essential to understand the changes imposed on mesocarnivores by urbanisation. The diet of water mongooses *Atilax paludinosus* has been studied in natural habitats of KwaZulu-Natal, South Africa. However, their urban conspecifics have been mainly overlooked. We investigated the feeding ecology by faecal analysis of water mongooses in the urban green space matrix of the Upper Highway Area of eThekwini, KwaZulu-Natal. Percentages of occurrence were used to analyse the dietary intake trends from 104 scat samples collected from 2018–2019. Urban water mongooses opportunistically consumed a wide diversity of prey items. However, their diet was dominated by three main categories (relative percentage of occurrence: crustaceans-35.7%, invertebrates 19.9%, small mammals 19.1% and other (25.2%). Seasonal variations were only present for crustaceans and invertebrates. We found the presence of anthropogenic waste (chicken bones, plastic particulates and cigarette butts) in scat samples. This indicated that urban water mongooses were supplementing their diet and foraging on anthropogenic waste. Our study highlighted the generalist and flexible feeding habits of water mongooses in an urban matrix.

**Keywords**: *Atilax paludinosus*, Urban matrix, diet, scat analysis, generalist, mongoose

The establishment of conservancies is a popular private land management instrument used by the public and civil society to protect wildlife in South Africa (Scholes 2002). The eThekwini biodiversity policy, including the Durban Metropolitan Open Space System (D'MOSS), has partnered with public and local conservancies to safeguard the remaining unique biodiversity in the region (Roberts et al. 2011). The protection of core conservation areas, which are linked by corridors to neighbouring green spaces act as steppingstones, allows species to move within the urban matrix of eThekwini Metropole (Zungu et al. 2020a; Zungu et al. 2020b). These green pockets are interspersed within dense residential areas and often traversed by major motorways (Streicher et al. 2021). Mammalian mesocarnivores are dependent on these natural refugia to persist in the urban landscape (Widdows et al. 2015; Zungu et al. 2020a).

Streicher et al. (2021.) documented that water mongoose *Atilax paludinosus* modified their spatial behaviour (smaller home ranges and higher population densities) in response to a constraining environment. The water mongoose, also commonly referred to as the marsh mongoose, is a solitary and nocturnal mammal (Maddock and Perrin 1993). Belonging to the Herpestidae family, water mongooses have an extensive geographic distribution ranging from sub-Saharan Africa, stretching into eastern, western and central Africa (Do Linh San et al. 2015). They are however absent from the arid regions of Namibia, Botswana and South Africa (Skinner and Chimimba 2005). They are a relatively large mongoose species (body mass between 3.0–5.5 kg and have a total length of ~1 m) with a short muzzle and robust head (Skinner and Chimimba 2005). Water mongooses are dependent on either fresh or estuarine riparian habitats (streams, rivers, vleis, marsh, swamps, impoundments/dams) in proximity to vegetative cover (Baker and Ray 2013). The unwebbed digits of water mongoose splay on a muddy and wet substrate, facilitating effective walking and hunting in its preferred habitat types (Skinner and Chimimba 2005).

The diet of water mongooses occurring in KwaZulu-Natal Province, South Africa, inland and coastal are characterised as being a relatively broad generalist and aquatic, predominantly comprising crustaceans. Three studies of their scats from the highlands (Rowe-Rowe 1978), lowlands (Maddock 1988) and northern coastal area (Whitfield and Blaber 1980) of KwaZulu-Natal, highlight the diversity and range of dietary items water mongoose consume (amphibians, mammals, birds, insects, vegetation, carrion etc.). Their flexible dietary habits enable the species to co-inhabit areas with other smaller carnivore species with limited competition or confrontation (Maddock 1988).

Although there is literature on the dietary ecology of water mongooses in South Africa the diets of urban populations have not previously been investigated. We used scat analyses to gain insight into the persistence of water mongoose in an urban environment and the potential use of anthropogenically derived resources to supplement their diet. We aimed to determine the diet of water mongooses in the urban setting of residential Kloof, eThekwini, KwaZulu-Natal. We predicted that water mongoose diet would be dominated by widely occurring freshwater crabs *Potamonautes sidneyi* and supplemented with other foods/dietary items.

We conducted the study in the Upper Highway Area of eThekwini Metropole on the east coast of KwaZulu-Natal (-29.772048, 30.830620). The western section of eThekwini encompasses the residential areas of Kloof, Hillcrest, Everton, Gillitts and Waterfall (Smith 2017). The region is characterised by mixed levels of urban development interspersed by natural green spaces and residential gardens (Municipality 2007; Roberts 2009). The vegetation in these green spaces is dominated by KwaZulu-Natal Sandstone Sourveld grassland, with patches of shrubland and indigenous riverine forest (Mucina and Rutherford 2006). Approximately two-thirds of the grassland biome has been historical transformed (forestry plantations, cultivation and urban development). The D'MOSS policy prevents the further anthropogenic conversion of green spaces, ensuring the preservation of wild flora and fauna

in this remaining residential matrix (Roberts 2009). The area receives summer rainfall (December–February) with a mean annual rainfall volume of 974 mm and temperature range of 13.9°C to 24.0°C (http://en.climate-data.org/location/27097/).

We collected water mongoose scat samples from latrine sites located during a concurrent telemetry study (Streicher et al. 2020) in the area from February 2018–November 2019. Latrines were commonly located close to streams and rivers. We identified scat samples according to shape, size (diameter and length) and smell, per the scat identification guideline in Stuart and Stuart (2013). Cape genet *Genetta tigrina* and Cape clawless otter *Aonyx capensis* share an overlapping distribution range in the study site and also defecate at latrines sites. The presence of freshwater crab shells and lack of fish smelling odour is indicative of water mongoose scats (Stuart and Stuart 2013). Samples that did not contain freshwater crab shell underwent additionally scrutiny to avoid the inclusion of scats from co–existing carnivore species. Scat samples from middens were individually bagged and assigned a sample number, date of collection and a global positioning system (GPS) geographical location (where possible) using a Garmin Etrex 10 (Garmin, Lenexa, Kansas, United States). The unavailability of a field site freezer meant each sample was sun-dried to remove the moisture to prevent bacterial contamination and thus possible further decomposition and degradation of the sample (Holland et al. 2003). In addition, scats were opportunistically collected by interested members of the Krantzkloof Conservancy. We provided volunteers with scat collection materials, an identification key and a user-friendly protocol for appropriate sample collection.

To process scats, we individually soaked samples in glass beakers of 250 ml of water to soften each sample before processing. Samples were subsequently washed through a fine-mesh sieve (~1 mm) under running water and finally air drying under an extraction fume hood.

To ensure only water mongoose diets were analysed, we processed only scats containing the distinctive banded hairs of our target species. We identified the remains using a

dissection microscope (E. Leitz Wetzlar, Germany) with × 10 magnification. We categorised food items found into nine groups: crustaceans, small mammals, birds, invertebrates, vegetation, fish, amphibians, reptiles and anthropogenic waste pollution. Small mammal (Perrin and Campbell 1980; Bowland and Bowland 1989) and invertebrate (Scholtz and Holm 2008) prey remains were further identified to order and species level using several keys. We spread hairs retrieved from scat samples in a Petri dish, and a few strands were randomly selected to adequately represent the entire sample (Ramesh and Downs 2015). Hairs of each of these samples were identified based on species unique microstructure and characteristics (shape and colour) (Perrin and Campbell 1980; Keogh 1985). We generated negative imprints of hair strands by placing a thin layer of transparent nail polish on microscope glass slides with hairs on it (De Marinis and Agnelli 1993; Ramesh and Downs 2015). We identified invertebrate prey samples using diagnostic fragments (wings, shells, mandible, and head capsules). Birds, vegetation, fish, amphibians and reptiles were not identified to lower taxonomic levels. The remains of the samples that were unidentifiable were recorded as unknown within the various categories. Anthropogenic waste pollution was classified and recorded.

We analysed water mongoose dietary composition using two complementary methods, namely; the percentage of occurrence (PO: (total scats in food group/total scats)×100), the relative percentage of occurrence (RPO: (total number of species or taxonomic group occurrence × 100)/ total occurrence of all prey items in scats). These dietary analysis methods should be considered with caution because they are prone to overestimation bias because of two factors: 1. successional scat collection after a single large meal, and 2. small commonly consumed prey items (invertebrates) (Atkinson et al. 2002). The two factors can cause overestimation bias because of the occurrence of common particles (Atkinson et al. 2002). However, this potential bias was minimised in this study by the considerable sample size and collection of scats from various sites (Widdows and Downs 2015).

We compared water mongoose total and seasonal dietary intake for the number of prey items, the relative percentage of occurrence of dominant food items, and relative percentage of occurrence of each food category consumed. The data were not normally distributed and therefore comparisons between total dietary components were conducted using Friedman ANOVA (IMB SPSS version 27). We used a non-parametric Kruskal-Wallis for seasonal dietary intake test (IMB SPSS version 27). We conducted additional comparative analyses following formulae provided in Krebs (1999), to interpret and compare the results for the relative percentage of occurrence between pairs of seasons. The comparative tests for the food categories were: 1. Shannon–Wiener diversity index (H′; range 0−2.196 for nine groups), 2. Evenness measure of representation (J′; range 0−1), 3. Levin's standardised dietary niche breadth (BA; range 0−1), and 4. Pianka's dietary niche overlap (α; range 0−1). Comparisons between pairs of seasons follow the criteria in Do Linh San et al. (2020).

A total of 104 water mongoose scats were collected for the seasons of 2018 and 2019. We collected the lowest number of scats in the summer ($n$ = 21) and the highest number in spring ($n$ = 33; Table 6.1). We identified a total of 16 taxa in the scats of water mongooses. These included four orders of invertebrates, seven orders of small mammals, vegetation, reptiles, amphibians, birds and fish (Table 6.2). There was a significant difference in the overall dietary components consumed by water mongoose (ANOVA, $\chi 2 = 302.63$, df = 8, $P < 0.05$). The presence of crustaceans dominated water mongoose scats with a relative percentage of occurrence of (mean $\pm$ SD) 35.8 ± 0.83 % (Figure 6.1). Other invertebrates accounted for 19.4 ± 5.19 % of their diet throughout the year followed by small mammals with 19.2 ± 0.41 % of their diet (Figure 6.1). The remaining 25.7 % comprised of birds, anthropogenic waste pollution, vegetation, fish, amphibians and reptiles in order of decreasing percentage of occurrence (Table 6.2 and Figure 6.1).

The relative percentage of occurrence for main dietary constituents for water mongoose across the seasons ranged from 30.1–41.0%, 15.0–25.6% and 14.5–29.0% for crustaceans, invertebrates and small mammals respectively (Figure 6.2). Seasonal variations in relative percentage of occurrence were significant for crustaceans (Kruskal-Wallis, $H = 48.13$, df = 3, $P < 0.05$) and invertebrates (Kruskal-Wallis, $H = 10.27$, df = 3, $P = 0.02$). Seasonal differences for small mammals were not significant (Kruskal-Wallis, $H = 4.57$, df = 3, $P = 0.21$). The highest relative percentage of occurrence for crustaceans was recorded in winter, 41.0 ± 1.41 % (Figure 6.2). The highest relative frequency of occurrence for invertebrates was recorded in autumn, 25.6 ± 8.0 % (Figure 6.2). All the remaining seasonal variations in relative frequency of occurrence for the six less consumed food categories were not significant for water mongoose in this study (Kruskal-Wallis; birds: $H = 4.31$, df = 3, $P = 0.23$, anthropogenic food waste: $H = 4.16$, df = 3, $P = 0.25$, vegetation: $H = 0.67$, df = 3, $P = 1.00$, fish: $H = 2.78$, df = 3, $P = 0.43$, amphibian: $H = 7.37$, df = 3, $P = 0.61$ and reptile: $H = 4.10$, df = 3, $P = 0.25$) (Figure 6.2) .

Small mammal remains were found in 56 scats. Constituents in 49 scat samples could be conclusively identified down to genus level for nine genera. *Rattus* was the most commonly occurring genus of small mammals (33%), followed by *Aethomys* (29%), *Rhabdomys* (10%), *Mastomys* (6%), *Otomys* (6%), *Grammomys* (4%), *Lemniscomys* (4%) and *Mus minutoides* (2%).

The yearly dietary diversity was intermediate for the Shannon–Wiener diversity index (1.72; $H'_{max} = 2.196$). However, representative evenness was relatively high (0.78) (Table 6.1). There was indication of slight seasonal variation in both the species diversity and evenness indices (Table 6.1). The highest species diversity and evenness was recorded in summer ($H' = 1.803$, $J' = 0.821$) and the lowest in spring ($H' = 1.618$, $J' = 0.736$) (Table 6.1). The yearly dietary niche breadth was of an intermediate level ($B_A = 0.438$) with slight seasonal variation

(Table 6.1). The highest dietary niche breadth was recorded in summer ($B_A$ = 0.516) and the lowest in winter ($B_A$ = 0.391). Dietary niche overlap between the seasons was high (> 0.95).

**Table 6.1.** The number of water mongoose (*Atilax paludinosus*) scats in the urban landscape of the Upper Highway Area of eThekwini Metropole, KwaZulu-Natal, South Africa, from 2018 to 2019 with corresponding seasonal dietary parameters using relative frequency of occurrence (RPO/100).

| Measure | Autumn | Spring | Summer | Winter | Year |
|---|---|---|---|---|---|
| Number of scats | 23 | 33 | 21 | 27 | 104 |
| Mean number of prey categories | 2.2 ± 0.8 | 2.1 ± 1.1 | 2.7 ± 1.2 | 2.3 ± 1.1 | 2.3 ± 1.1 |
| Mean number of prey items | 2.1 ± 0.9 | 2.2 ± 1.0 | 3.0 ± 1.1 | 2.4 ± 0.9 | 2.4 ± 1.0 |
| Shannon–Wiener diversity index (H′) | 1.63 | 1.62 | 1.80 | 1.65 | 1.72 |
| Evenness of representation (J′) | 0.74 | 0.74 | 0.82 | 0.75 | 0.78 |
| Standardised dietary niche breadth (BA) | 0.39 | 0.41 | 0.52 | 0.39 | 0.44 |

**Table 6.2.** Prey items recorded in the scats of water mongoose (*Atilax paludinosus*) in the urban landscape of the Upper Highway Area of eThekwini Metropole, KwaZulu-Natal, between 2018 and 2019 (PO = percentage of occurrence, RPO = relative frequency of occurrence)

| Food item | PO | RPO |
|---|---|---|
| **Crustaceans** | **81.7** | **35.7** |
| **Invertebrates** | **46.7** | **19.9** |
| Coleoptera | 9.2 | 4.0 |
| Dictyoptera | 1.8 | 0.8 |
| Hymenoptera | 2.8 | 1.2 |
| Unknown arthropod | 31.2 | 13.7 |
| **Rodents** | **44.8** | **19.1** |
| *Rattus* spp. | 10.1 | 4.4 |
| *Mastomys* spp. | 2.8 | 1.2 |
| *Dendromus* spp. | 2.8 | 1.2 |
| *Mus minutoides* | 0.9 | 0.4 |

| | | |
|---|---|---|
| *Aethomys* spp. | 12.8 | 5.6 |
| *Grammomys* spp. | 1.8 | 0.8 |
| *Otomys* spp. | 2.8 | 1.2 |
| *Rhabdomys* spp. | 3.7 | 1.6 |
| Unknown rodent | 6.4 | 2.8 |
| **Vegetation** | **15.6** | **6.8** |
| **Reptiles** | **0.9** | **0.4** |
| **Amphibians** | **1.8** | **0.8** |
| **Birds** | **19.3** | **8.4** |
| **Anthropogenic waste pollution** | **15.6** | **6.8** |
| **Fish** | **4.6** | **2.0** |

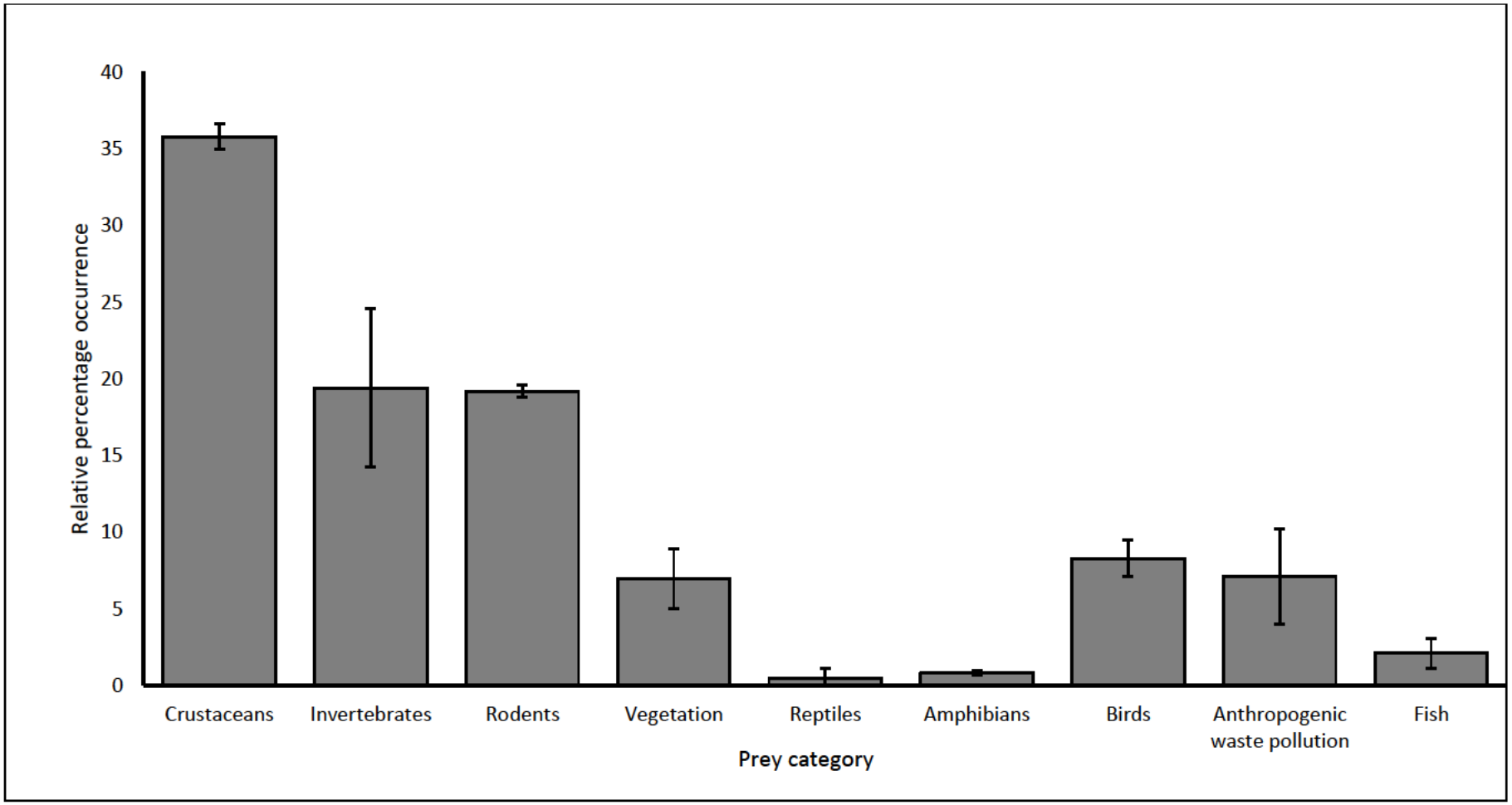


**Figure 6.1.** The annual mean (± SD) relative percentage of occurrence of prey categories from the scats of water mongoose (*Atilax paludinosus*) in the urban landscape of the Upper Highway Area of eThekwini Metropole, KwaZulu-Natal for seasons of 2018 and 2019.

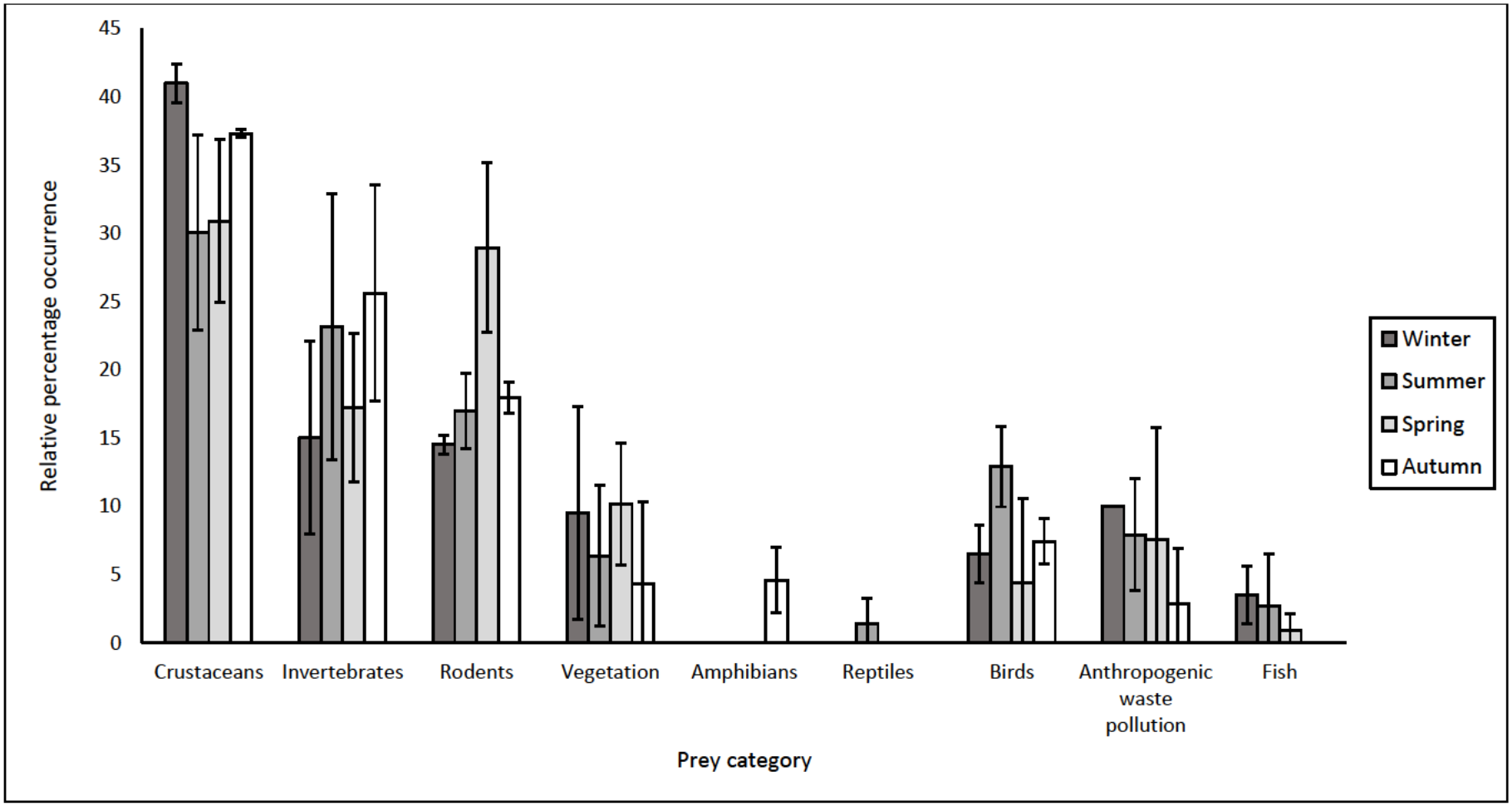


**Figure 6.2.** The mean (± SD) seasonal variation in the relative percentage of occurrence of prey categories from the scats of water mongoose (*Atilax paludinosus*) in the urban landscape of the Upper Highway Area of eThekwini Metropole, KwaZulu-Natal for seasons of 2018 and 2019.

The water mongoose is described as an opportunistic and generalist carnivore (Skinner and Chimimba 2005; Do Linh San et al. 2020). Past studies in non-urban areas have found that their diet is dominated by aquatic prey (crabs, amphibians, fish and aquatic invertebrates) (Rowe-Rowe 1978; Maddock 1988). Other terrestrial species frequently occur in their diet; including rodents, birds, reptiles, and invertebrates (Skinner and Chimimba 2005). Water mongooses have been documented to scavenge in carrion when presented with the opportunity (Maddock 1988). The dietary flexibility of water mongooses typically allows them to shift to secondary prey items (mammals and arthropods) on the absence of their preferred prey, consuming a high number of food groups particularly in inland areas (Angelici 2000; Do Linh San et al. 2020). Overall, they show remarkable flexibility in their dietary habits.

Similarly, in our study, scat analyses showed that the urban water mongoose was a generalist in its feeding habits with a diverse array of prey consumed. The overall diet was dominated by three preferred groups; crustaceans, invertebrates and rodents (Table 6.2, Figure 6.1). Previous literature shows that crustaceans are the dominant and primary food type, when available in freshwater systems (Rowe-Rowe 1978; du Toit 1980; Louw and Nel 1986; Maddock 1988; Baker 1989). Moreover, there were seasonal differences in the consumed prey items: crustaceans (*Potamonautes* spp.) and invertebrates, but small mammals in this study. Crustaceans and invertebrates were consumed at higher rates during winter and autumn seasons, respectively, and lower rates in summer and winter, respectively. Other secondary prey items frequently occurred throughout the seasons in water mongoose scats and showed no seasonal difference. This fact highlights that certain prey items were positively selected throughout the seasons with other secondary prey items fed upon opportunistically when encountered (Baker 1989).

Furthermore, the subtropical climate the area experiences may provide a possible explanation for why an array of prey species were available throughout the seasons as was

evident from the water mongoose scat samples. Dietary breadth and evenness were highest during the summer months. Urban water mongooses diets did not show the wide dietary diversity and range as found in inland studies (Do Linh San et al. 2020), which further highlighted that preferred prey items were seasonally abundant. Interestingly, amphibians were poorly represented in the diets of water mongooses in this study compared with previous literature (Maddock 1988; Skinner and Chimimba 2005; Do Linh San et al. 2020).

Although our results showed that crustaceans were the primary prey group for urban water mongooses, the relative frequency of occurrence calculations are prone to overestimation of small prey items and prey with undigestible parts (Klare et al. 2011). A biomass index is considered to be a more accurate measure of dietary composition (Klare et al. 2011). However, the percentage of occurrence estimators were suitable indexes for the main aims of this study on urban water mongooses. Furthermore, no previous studies have examined urban water mongoose dietary habits, and the proclivity of multiple individuals to use a single midden site precluded estimation of biomass from this study (Widdows and Downs 2015).

There is a global concern around the presence of anthropogenic waste pollution in scats of urban mammals. Our study indicated that urban water mongooses might frequently be ingesting non-target anthropogenic food sources (relative percentage of occurrence: 6.9%) to supplement their diet, particularly in the winter months (relative percentage of occurrence: 10.0%) (Figure 6.2). The crepuscular nature of water mongooses may facilitate scavenging in refuse bins during the meagre months and when human activity is minimal. Additionally, scat samples collected from public spaces (parking lots and picnic sites) had higher levels of anthropogenic waste (chicken bones, plastic and cigarette butts) as compared with scats collected in Krantzkloof Nature Reserve and D'MOSS areas. Similar feeding plasticity in feeding behaviour, especially in winter and feeding on anthropogenic waste, was documented for the cape genet (Widdows and Downs 2015, Present study) in the same region. The

supplementation of mammalian diets with anthropogenic waste, especially in urban areas, is not unique to these studies (Widdows and Downs 2015; present study). Mesocarnivores globally (red fox: *Vulpes vulpes*, coyote: *Canis latrans*, raccoon: *Procyon lotor*, stone marten: *Martes foina*, banded mongoose: *Mungos mungo*, striped skunk: *Mephitis mephitis*, European badger: *Meles meles*, and dingo: *Canis dingo*) have been documented to supplement their diet with anthropogenic waste in urban areas (Harris 1984; Clark 1994; Gilchrist and Otali 2002; Contesse et al. 2004; Eskreys-Wójcik and Wierzbowska 2007; Morey et al. 2007; Murray et al. 2015). The enticement of human refuse is suggested to have a habituation effect on mammalian carnivores and is consequently reducing the proximity between mesocarnivores and humans (Bateman and Fleming 2012). The subsequent diminution of the segregating boundary could therefore increase the rate of negative human–wildlife interactions.

The diversity of prey items found in the diets of urban water mongooses highlighted the broad species dietary breadth and generalist feeding habits of this urban population. Crustaceans were the most dominant prey type in this study; however, their opportunistic and flexible feeding behaviour underscored the species ability to shift dietary habits to capitalise on prey items based on abundance, location and season. The evident presence of anthropogenic waste pollution in water mongoose scats may be aiding the species ability to persist in natural fringe areas in an urban mosaic habitat. Our study further emphasised the importance of responsible anthropogenic refuse disposal. Future studies should incorporate faecal DNA identification techniques to reveal precise feeding strategies by urban water mongooses. Lastly, our study confirmed the need to understand the ecology of urban mesocarnivores because of the general lack of information available in human–dominated systems.

## 6.2 Acknowledgements

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# CHAPTER 7

# Conclusions

## 7.1 Introduction

In this final chapter, I summarise and discuss the main findings of my study. In line with the research aims and objectives in Chapter 1, I also present management recommendations, and future directions for research on the three members of the Herpestidae family (large grey: *Herpestes ichneumon*, water: *Atilax paludinosus* and white-tailed mongoose: *Ichneumia albicauda*; - with particular emphasis on the water mongoose).

Human population growth and the concomitant increase in anthropogenically modified landscapes (agricultural and urban) are threatening biodiversity and ecosystem services (Vitousek *et al.*, 1997; Chapin *et al.*, 2000; Foley *et al.*, 2005). Human population growth is responsible for the overexploitation of natural resources, the degradation of ecosystem functions, land fragmentation and depletion of available resources for wildlife (Vitousek *et al.*, 1997; Altieri, 1999; Chapin *et al.*, 2000; Foley *et al.*, 2005). With natural landscapes being radically altered, the IUCN conservation status of many species remains unknown because of insufficient research. Mesocarnivores are integral to ecosystem function and structure through dynamic top-bottom regulation (Prugh *et al.*, 2009; Roemer, Gompper & Van Valkenburgh, 2009; Brashares *et al.*, 2010). However, a knowledge gap exists in the ecology of many mesocarnivores in southern Africa. This knowledge gap is illustrated in the number of available published articles on the spatial ecology of mongooses (Chapter 1). Our four field-based research objectives were initially developed to address the evident knowledge gap, particularly with anthropogenic land-use change, in the ecology of the above-listed three mongoose species. The primary goals were to focus on the mongooses inhabiting fragmented natural habitat farmland mosaic in the Midlands of KwaZulu-Natal (KZN), South Africa, and then contrast this with mongoose populations in the urban residential mosaic of the Upper Highway Area of

KZN (Chapters 3 and 4). It was only possible to complete the urban aspect of this study on water mongoose. Although large grey and white-tailed mongooses were historically abundant, these species were absent from the urban study sites we focussed our fieldwork in (Rowe-Rowe, 1978).

The spatial ecology of the Herpestidae family has been poorly studied within southern Africa, and most knowledge is anecdotal. In KZN, there has been limited progress in the understanding of the ecology of the family as a whole. Increased anthropogenic fragmentation of landscapes and the absence of predatory control may aid the range expansion of several Herpestidae members (Prugh *et al.*, 2009; Roemer, Gompper & Van Valkenburgh, 2009). However, the lack of research on smaller carnivores continues to be a concerning factor, particularly in a rapidly changing world.

Although categorised by the IUCN as 'Least Concern', the status of the three mongoose species; large grey, water mongoose and white-tailed mongoose, in KZN and southern Africa remains unclear. Furthermore, there have been limited contributions to the understanding of their spatial ecology since the publications of Rowe-Rowe, Maddock and Perrin in the late 1980s (Maddock, 1988; Rowe-Rowe, 1992; Skinner & Chimimba, 2005). Several mesocarnivore species are known for their behavioural adaptability and plasticity (Sih, Ferrari & Harris, 2011; Lowry, Lill & Wong, 2013; Widdows & Downs, 2015, 2016). However, because of their crepuscular nature and diminutive size, there is no clear understanding of how anthropogenic land-use change and fragmentation are affecting their behaviour (Maddock, 1988; Rowe-Rowe, 1992; Skinner & Chimimba, 2005; Ramesh & Downs, 2014). Insight into how these three species of mongoose interact with the surrounding fragmented landscape (rural and urban) contributes essential information on their ecology. This study has contributed substantially to enhancing the previously negligible spatial ecological knowledge available on the three mongoose species. The research presented here has also added valuable information

into the effects of the continued anthropogenic transformation (Rural–Urban), which was previously unknown.

### 7.2 Research findings

Our review of urban mesocarnivores allowed us to evaluate global trends and knowledge gaps that persist globally. Studies on urban mammalian carnivores have increased rapidly in the past two decades (Chapter 2), representing a general expansion of the field and surge in research interest (Magle *et al.*, 2012). Our review reveals a bias in the research targets when urban mesocarnivore studies have been conducted (Chapter 2). The majority of studies in urban areas has focused on several mesocarnivore species belonging to North America (e.g., *Canis latrans*) and Europe (e.g., *Vulpes vulpes*). This is despite Africa, Asia and South America supporting the majority of mammalian mesocarnivores species (McDonald, Kareiva & Forman, 2008) and that these areas are rapidly becoming urbanised. The conspicuous exclusion of urban mesocarnivore research in developing countries requires addressing to prevent further lags in global conservation efforts.

The second objective was to use Global Positioning System (GPS) telemetry data to investigate the home range of the three mongoose species (large grey, water and white-tailed) in the KZN Midlands natural habitat and farmland mosaic landscape (Chapter 3). This study is the first to provide evidence for the home range requirements of the three species using GPS telemetry in the KZN Midlands. GPS data indicated that the home range sizes for white-tailed mongoose (Chapter 3) were similar to those found in past studies (Admasu *et al.*, 2004; Skinner & Chimimba, 2005). As a consequence of their relatively small home ranges, white-tailed mongooses obtained their necessary resources without extensive foraging ventures. In contrast, the home range size estimations obtained in the present study for the telemetered large grey and water mongoose species (Chapter 3) were substantially larger than presented previously

(Maddock, 1988; Skinner & Chimimba, 2005). The previously recorded small home range sizes could be attributed to two factors. Firstly, there has been an improvement in tracking technologies through GPS tracking, and these technological improvements have allowed for continuous and accurate monitoring of collared individuals, even in relatively hostile environments (Cooke *et al.*, 2004; Kays *et al.*, 2015). GPS tracking technology, therefore, provides a more accurate representation of the spatial ecology of its collared individuals. Secondly, the increase in anthropogenic fragmentation of landscapes subsequently reduces natural areas available for wildlife. Landscapes have become dominated by expansive levels of monoculture farming, with intermittent pockets of refugia scattered around agricultural landscapes (Streicher, Ramesh & Downs, 2020). Individuals, therefore, may be forced to increase their foraging expeditions, and thus their home ranges, to obtain resources from these intermittent pockets. Although the three species are characterised by having generalist diets and can switch between available resources, they still show dietary preferences (Maddock, 1988; Baker, 1989; Rowe-Rowe, 1992; Skinner & Chimimba, 2005; Do Linh San *et al.*, 2020). These dietary preferences enable each species to occupy a differential niche and thereby reduce the potential for interspecific competition when co-inhabiting areas (Chapter 3).

In line with our third objective, we also used GPS telemetry data to investigate the home range and habitat use of the water mongooses in the urban residential matrix of the Upper Highway Area, eThekwini Municipality, KZN (Chapter 4). This study is the first to provide evidence of spatial ecology using GPS telemetry in an urban setting for this species. Home range sizes for urban water mongooses illustrated individual variation; however, they were constricted to natural refugia (Durban Metropolitan Open Space Systems (D'MOSS)) within the mosaic (Chapter 4). Water mongooses appeared to have high population densities in these "Green Spaces" within the urban mosaic, avoiding built-up urban areas in favour of fragments of forest and bushland habitat types close to natural water.

In terms of behaviour, we found there were high levels of spatial overlap between females, and female-with-male water mongooses (Chapter 3). Male water mongooses displayed territorial behaviour with limited spatial overlap between other urban male mongooses despite their restricted home ranges. The home range size for all three home range estimation measures (MCP, KDE and LoCoH) were significantly smaller than results obtained in the KZN Midlands natural habitat and farmland mosaic study (Chapter 3). The study accentuates the species' general adaptive nature which enables it to persist in an urban residential matrix. However, urbanisation is a significant threat to the species because they rely upon the D'MOSS green spaces for essential resources. Water mongooses are unlikely to become true urban exploiters, but instead, we expect them to persist in the remaining isolated natural fragments of natural refugia in and on the edge of the urban matrix landscape.

Our online survey questionnaire enabled us to assess public perceptions and threats that mammalian mesocarnivores face across a land-use gradient (rural–urban). Our multi-species approach in the survey (Chapter 5) allowed us to reveal the discrepancy in how species are impacted disproportionately by Human-Wildlife Interactions (HWI). We find that the "one size fits all" approach is no longer holds for wildlife conservation (Ripple *et al.*, 2017; Trajçe *et al.*, 2019). A wide variety of different opinions were held by participants that reside in rural or urban landscapes—the majority of respondents from both land-use types considered medium-sized mammalian carnivores not to be threatening and are indeed vital for the environment (Chapter 5). However, not all mesocarnivores were perceived equally: black-backed jackal (*Canis mesomelas*) were often singled out as a problematic pest species by rural respondents. Consequently, black-backed jackals face additional anthropogenic pressures of targeted human persecution. Smaller and more behaviourally flexible mesocarnivores (mongoose species and cape genet *Genetta tigrina*) were the most sighted species in both land-use types indicating a

potential population increases through "mesocarnivore release" (Roemer, Gompper & Van Valkenburgh, 2009).

Moreover, we were able to identify that collisions with vehicles were the primary cause of fatality in mesocarnivores in both land-use types (Chapter 5). By comparing rural and urban HWI with mammalian mesocarnivores along a land-use gradient allowed us to recognise trends in perspectives and threats. Management practitioners can acknowledge these similarities and equip themselves in establishing implementable strategies to safeguard mesocarnivores.

Urban water mongoose dietary composition was broad but was dominated by three major food categories (crustaceans, invertebrates and rodents) (Chapter 6). Previous literature shows that crustaceans are the dominant and primary food type when available in freshwater systems (Rowe-Rowe, 1978; du Toit, 1980; Louw & Nel, 1986; Maddock, 1988; Baker, 1989). The generalist dietary habits of urban water mongooses allow them to exploit resources that are readily available in the urban environment (Chapter 6). Water mongooses' ability to easily exploit available resources in a variety of habits, supplements current literature on the species' dietary flexibility (Rowe-Rowe, 1978; Maddock, 1988; Do Linh San *et al.*, 2020). Seasonal variations in mongooses dietary intake were only present for crustaceans and invertebrates. The subtropical climate the study site experiences may be a possible explanation for the potential lack of seasonal variation in diet. Evidence of the use of anthropogenic resources (cigarette butts, plastic and old chicken bones) was present in scats of urban water mongoose. The supplementation of water mongoose diets with anthropogenic resources was highest during the winter months (Chapter 6), indicating the species resourcefulness during lean months, but also the lack of natural dietary resources in this modified environment. Similarly, urban cape genets showed increased use of anthropogenic resources in winter (Widdows & Downs, 2015).

### 7.3 Future research work

Our study has answered several questions and raised further questions that we recommend future conservation approaches to research.

Confirmation of the population levels and whether these mesocarnivore species are undergoing "mesocarnivore release", would be a necessary first step in further research. Following this establishing how mesocarnivore release may respond differently across a landscape gradient. Once the population dynamics of these species are better understood, a subsequent investigation would have to be conducted into the impacts of mesocarnivore release on ecosystem function, structure, and how ecosystem dynamics change as a result. The spatial ecology of urban residing large grey and white-tailed mongooses requires investigating. There have been increased sightings of these species in urban areas of uMgungundlovu, where the habitat is more favourable for these two species of mongoose compared with the residential mosaic of the Upper Highway Area. Additionally, caracals (*Caracal caracal*), appear to be expanding their range into green spaces of the residential mosaic of the Upper Highway Area. Future research should investigate the factors influencing this species expansion into the Eastern Seaboard as well as explore the spatial ecology of the species in an urban context.

Further research on the genetics and health of these mongoose species is necessary for understanding how the populations' genetic variation is responding to the change in population dynamics, as well as how this impacts their health and ability to persist in the agricultural and urban matrix. The spatial configuration of natural habitat patches is an essential factor for terrestrial mammal species richness and diversity. Future studies should aim to establish the effects of landscape connectivity between natural patches on mesocarnivores in both rural and urban landscapes in KZN. Aside from hearsay, there appears to be a gap in information on the prevalence of zoonotic diseases, pathogens and transmitting of these diseases, as well as

whether mesocarnivores experiencing anticoagulant poison exposure are associated with continued anthropogenic land-use change.

Lastly, there is a need to increase urban ecological research output on mesocarnivores in South Africa and Africa. There is a disproportionate representation of mesocarnivore urban ecological research Europe and Northern America (Streicher unpublished data). These studies have extensively focused on specific species, leaving the majority of the worlds urban (developing countries) mesocarnivores relatively unstudied.

### 7.4 Concluding remarks

This dissertation provides important insight on the ecology of three solitary mongooses (large grey, water and white-tailed mongoose) in the fragmented natural farmland and water mongoose in the urban mosaic of the KZN, South Africa. General behavioural plasticity and adaptive nature of Herpestidae are represented by the species ability to co-exist in fragmented natural farmlands and by urban water mongooses in modifying their behaviour (spatial and diet) in attempt to persist in these changing landscapes.

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